Nutrient Inputs and Trophic Status of Carlingford Lough

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Executive Summary

Carlingford Lough is a coastal embayment and supports a wide range of aquaculture and fishing interests, much of which is protected by a range of conservation schemes. With regards to this, the aims of this project were to investigate whether eutrophic conditions currently exist in Carlingford by the collection of water quality and hydrographic data, and the calculation of nutrient loads to the system.

Direct inputs of nutrients are predominantly to the inner Lough. On an annual basis the Newry River contributes 84, 24, and 57 % of the total loads of NO_3, NH_4 and SRP respectively, and is clearly the most important source of nutrients. STWs are the largest source of NH_4 although ammonium contributes < 2 % of the total load of DIN. The load of NO_3 is seasonal with the load in winter months typically being > 100 T month^{-1}, but during the summer loads are generally reduced to < 50 T month^{-1}.

Freshwater discharge from the Newry River did affect the salinity of the Lough, although generally it was well mixed both vertically and horizontally. Nutrient concentrations in the Lough decreased on moving seawards, due to the major sources discharging at the head of the Lough, and subsequent dilution with nutrient poor water from the Irish Sea. Nutrients showed the characteristic seasonal patterns of summer minima and winter maxima found in temperate waters. Following the initial spring bloom of phytoplankton, NO_3 concentrations were generally reduced to < 1 μM in the mid and outer Lough until September and the N:P ratio fell below 15. At some stations in the inner Lough, concentrations of NO_3 were higher than the mid and outer during the summer probably due to a combination of the nutrient sources discharging to this zone and remineralization of nutrients in the shallow water and sediments.

The phytoplankton population was dominated by diatoms, but dinoflaggelates contributed significantly in June. Throughout the Lough chlorophyll concentrations increased from winter minimum concentrations to between 5 and 10 μg l^{-1} during March and April. Following this initial 'bloom', the stations could be broadly grouped into two according to the trends observed. Seaward of station 10 chlorophyll concentrations decreased to < 3 μg l^{-1}, whereas concentrations were sustained at > 3 μg l^{-1} at stations landward of station 10. The evidence presented suggests that N is the

primary limiting factor for phytoplankton growth, and therefore the sustained levels of chlorophyll in the inner Lough are probably caused by continued inputs of nutrients to this zone. There is no evidence to suggest that the Lough is currently eutrophic.

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Chapter 1. Introduction.

1.1 The study area

Carlingford Lough is the most southern of the five sea loughs around the coastline of Northern Ireland (Fig. 1.1).

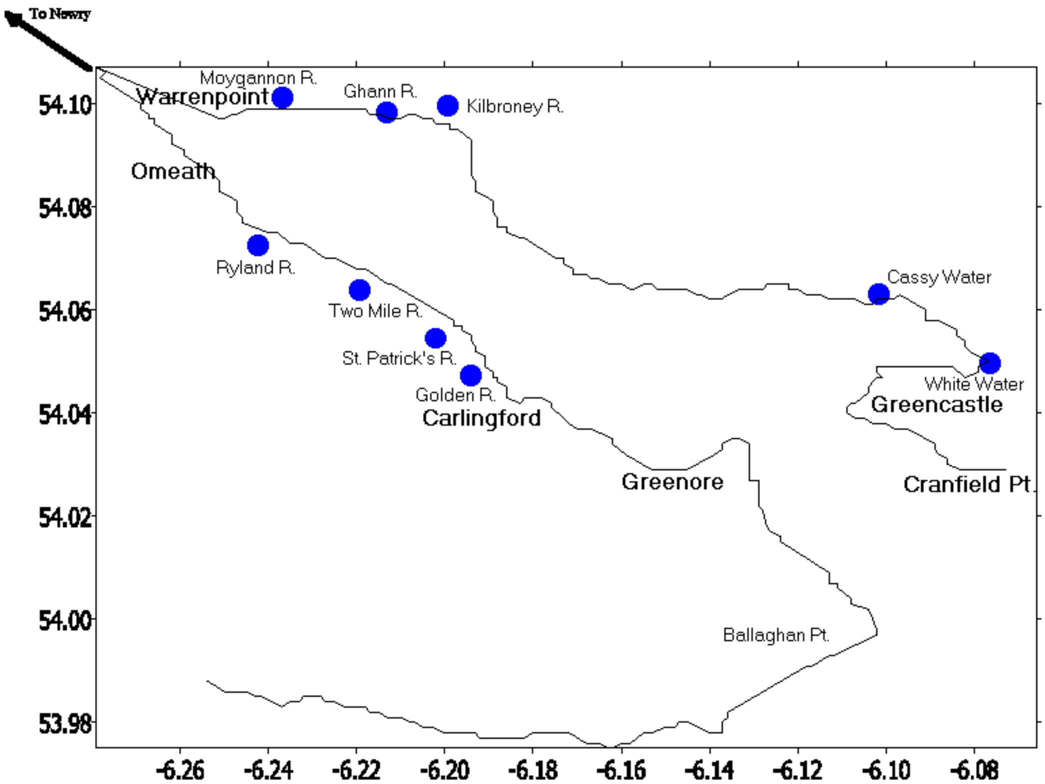

Figure 1.1 Map showing position and main features of Carlingford Lough.

It is a cross-border water body, bounded by County Down and County Armagh in Northern Ireland and by County Louth in the Republic of Ireland. Carlingford Lough supports a wide variety of aquaculture and fishing interests, including cultivation of oysters, mussels and clams as well as lobster and crab potting. To date, studies focusing on Carlingford Lough have largely centred around the use of the Lough as a major site for mariculture, and on sewage disposal activities in the Lough. The aims of this project were to investigate whether eutrophic conditions currently exist in Carlingford Lough or may be expected to develop if action to reduce nutrient levels is not taken in the future. This was determined by the collection of water quality, primary productivity and hydrographic data, and by the calculation of estimates of nutrient loads into the Lough.

1.1.1 Physical characteristics

Carlingford Lough extends in a northwest - southeast direction from the Clanrye or Newry River which is the major freshwater source, to the shallow mouth which is guarded by a chain of limestone islands.

Table 1.1 Physical characteristics of Carlingford Lough.

Total Area	Intertidal Area	Length	Max. Depth	Max. Width	Coastline	LW Volume
km^2	km^2	Km	m	km	km	10^6 m^3
51.4	14.9	16.5	36.0	5.5	63.4	146.4

The Lough is generally shallow with the average depth between 2 and 10 m, although the narrow channels that run along the centre of the Lough may be as deep as 25 m (Fig. 1.2).

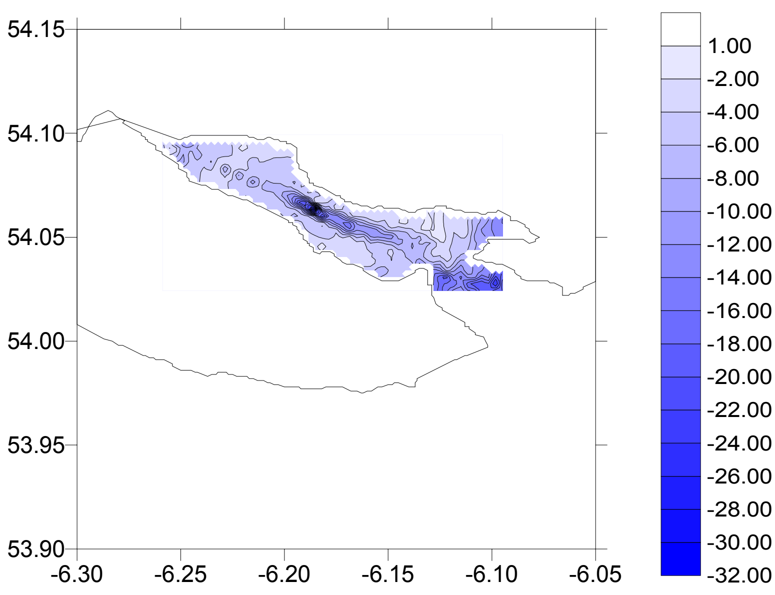

Figure 1.2 Contour plot of depth from RoxAnn survey

At low-tide, an extensive area (14.9 km^2) of intertidal area is exposed. The upper reaches of the Lough are extremely narrow due to land claim occurring south of Newry. The greatest tidal movements occur in the narrow channels that run along the centre of the Lough.

The upper reaches of the Lough are shallow and are dominated by a fine muddy sand bed while the substratum at the entrance of Carlingford Lough is mostly boulder and cobble, mixed with patches of bedrock (Fig. 1.3). The sediments in the centre of the Lough are widely varied.

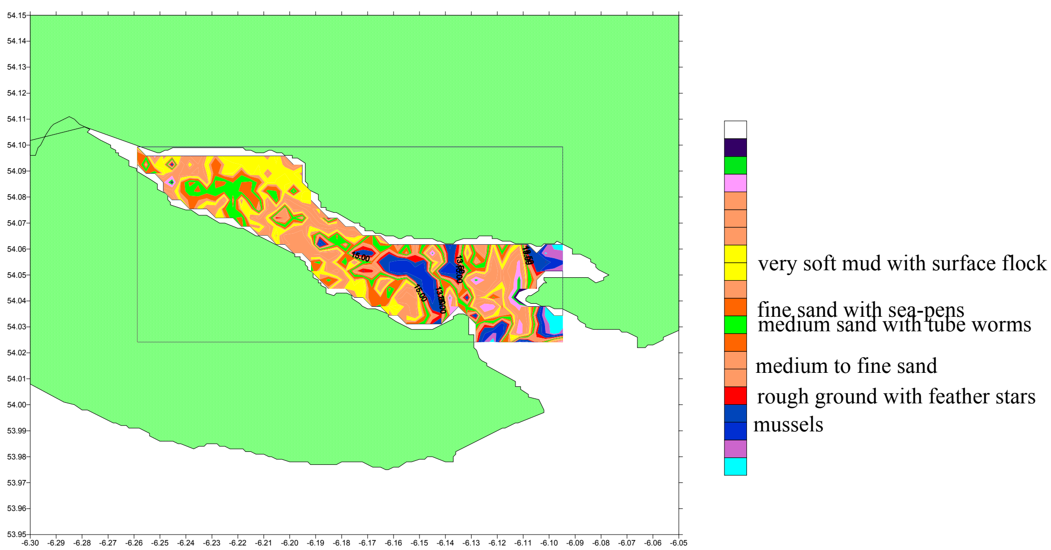

Figure 1.3 RoxAnn classification of sea bed types.

1.1.2 Biological characteristics

Carlingford Lough is host to aquatic estuarine fauna of considerable interest including extensive beds of sea-pens (*Virgularia mirabilis*) at the mouth of the Lough, sand and rock reefs in the central section and fast-water communities near the mouth. The Lough supports a number of noteworthy marine species, some of these being warm-water species which in Northern Ireland, are found only at this location.

Towards the mouth of the Lough, there are small areas of saltmarsh between Carlingford and Greenore, and at Mill Bay where the White Water River flows into the Lough. Towards the land, there is a transition in the vegetation from saltmarsh to fen and in certain areas, the saltmarsh is showing symptoms of erosion.

The Lough supports an internationally important population of wintering light-bellied brent geese (*Branta bernicla hrota*), and nationally important populations of eight species of wintering waterfowl. Green Island, in the lower reaches of the Lough, supports important breeding populations of terns. A small population of common seals breed within the Lough, and grey seals feed within the outer parts of the Lough.

1.1.3 The Carlingford catchment

Carlingford Lough drains an area of 474 km^2, encompassing both Northern Ireland and the Republic of Ireland. The largest town within the catchment is Newry (41,569 inhabitants), on the Clanrye River (Fig. 1.1). There are three small towns on the County Down side of the Lough, namely Warrenpoint, Rostrevor and Greencastle. On the County Louth side, Greenore, Carlingford and Omeath are the main towns.

CORINE land cover data indicate that the land in the immediate vicinity of the Lough supports a mixture of forest, rough and improved grazing and small areas of agricultural land. On the northern shore, the agricultural land is confined to an area behind Killowen Bank and another area just seaward of the start of the Newry Canal while on the southern shore, agricultural land is confined to an area around Omeath. Coniferous forest is found in the area east of Rostrevor Bay, and also in a band behind the rough pasture which is found running down the south side of the Lough, from Omeath to Carlingford. Mixed forest is found adjacent to the Newry River, inland of Narrow Water. Pasture for the grazing of mainly sheep and dairy cows is found north of Rostrevor Bay, and also in the area around Warrenpoint, and on the south side of the Newry Canal. Rough grazing for sheep is found adjacent to the Lough between Omeath and Carlingford. Further inland, the area is dominated by a mixture of pasture, natural grassland, arable, moors and heathland.

1.2 Uses of the Lough

1.2.1 Marine fisheries and shellfisheries

Historically, Carlingford Lough was renowned for its herring fishery, but in the 19th century, the fishery within Carlingford collapsed as numbers of herring dropped dramatically. Today, herring fishing activities are confined to sporadic efforts during summer to catch shoaling herring. Only a small number of boats engage in the herring fishery, operating outside the Lough mouth.

Similarly, the native oyster fishery (*Ostrea edulis*) collapsed in 1845 due to combined overfishing and exploitation of juveniles for reseeding other areas. The fishery recovered to some extent in the early 1900's, but collapsed once again, due to overfishing and silting in the Narrow Water area, as a result of the development of Newry and Warrenpoint. An effort was made to revive the industry in the early 1970's by Bord Iascaigh Mhara (BIM) but trials with the Pacific oyster (*Crassostrea gigas*) were more successful and Carlingford Lough was selected by BIM to cultivate the Pacific oyster.

Currently, mariculture is an important industry within Carlingford Lough, particularly in the large intertidal areas. In the Northern Ireland side of the Lough, there are six mariculture sites, extending from Warrenpoint to just seaward of Green Island at the mouth of the Lough (B. Magorrian, Fisheries Division, DANI, pers. comm.). In the North, Pacific oyster trestle cultures are located south of Greencastle and north of Mill Bay. Five Pacific oyster units operate in the Republic of Ireland side of Carlingford Lough, with a combined estimated annual harvest of 400 tonnes, the majority of which are exported to France. Mussel (*Mytilus edulis*) culture, including raft and bottom culture systems, operate at Greencastle Point, Mill Bay, Killowen Point and Rostrevor Bay and on several farms in the Republic of Ireland. Wild mussel dredging also occurs within the Lough, with approximately 1,000 tonnes removed per annum (1992 data; Ball *et al*, 1997). Manila Clams (*Tapes semidecussata*) are cultured in the north of Mill Bay and raft culture of scallops (*Pecten maximus*) is confined to a site off Greencastle Point.

1.2.2 Effluent disposal

The main sewage treatment works in Northern Ireland that discharge into Carlingford Lough are located at Newry Town (in the Newry River estuary) with a population equivalent (PE) of 41,569, Warrenpoint with a PE of 13,887 and Cranfield, where the presence of a tourist facility results in a variable PE between 400 and 4,000. Preliminary treatment is carried out at Cranfield, secondary at Warrenpoint and treatment at Newry is currently primary, although a new works is to be completed in the near future, and will carry out at least secondary treatment of sewage. All sewage discharged into Carlingford Lough from the Republic of Ireland is untreated, except at Carlingford where a new treatment works will open in 2000. The design capacity of this works will be 1,400 PE, compared to the current discharge of 550 PE, and will carry out secondary treatment (D. O'Neill, Louth County Council, pers. comm.). Omeath and Greenore sea-outfalls currently have a PE of 255 and 143, respectively.

1.2.3 Nature conservation

Much of Carlingford Lough's northern foreshore lies within an Area of Outstanding Natural Beauty (AONB). The AONB encompasses 56 kilometres of coastline, including Carlingford Lough itself, together with extensive stone walls and settlements fringing 700 metre granite peaks, moorland and a deep valley of the central core of the Mourne Mountains. This coastal plain is also part of the Mourne and Slieve Croob Environmentally Sensitive Area (ESA), an area roughly concomitant with the Mourne AONB, but excluding the high mountain peaks. The Lough, including Green Island, also qualifies as a Ramsar site, and as a Special Protection Area (SPA). Greencastle Point Islands as well as Green and Blockhouse Islands are RSPB reserves. Rostrevor Forest, on the inner north shore of the Lough, is a National Nature Reserve (NNR). In the Republic, the southern shore of Carlingford Lough is a Special Protection Area (SPA), and this forms part of a larger area that is proposed as a Natural Heritage Area and also as a Special Area of Conservation (cSAC).

1.2.4 Navigation

There is limited industry around Carlingford Lough. Greenore and Warrenpoint are important local ports and the latter developed at the expense of the former port at Newry which closed in 1974 due to the limited size of ship which could be accommodated in the Newry Canal.

1.3 Previous studies of the Lough

The most recent survey work on Carlingford Lough, aimed at assessing the impact of expanding bivalve culture on the ecosystem of the Lough, was carried out between 1992 and 1993 by Ball *et al* (1997). There was little evidence of vertical stratification within the Lough, due to shallow depths combined with a large tidal range, strong currents and a relatively small freshwater input. The Clanrye River was calculated to be responsible for approximately 70% of the total freshwater and freshwater-derived nutrient input into the Lough. River nitrate concentrations were not seasonal, ranging between 37 µM and 57 µM. Soluble reactive phosphorus (SRP) in the Clanrye River varied from 5.5 µM to 18.9 µM, with highest values recorded during the summer. Within the Lough, all nutrients exhibited the characteristic seasonal pattern of winter maxima and summer minima. The Lough receives dissolved nitrogen from both the Irish Sea and from rivers (Ferreira *et al*, 1998). Nitrate concentrations here ranged between <0.6 µM and 36 µM, with highest values recorded around Omeath. Balls *et. al.* (1997) noted that during late spring and summer, SRP levels fell to below 2 µM but at no stage was there complete depletion of this nutrient. A similar pattern was observed for silicate, with levels related closely to salinity. The concentration of suspended particulate matter in the Lough ranged between 5 mg m^{-3} and 66 mg m^{-3}, with the highest values found at the Omeath site. Particulate organic carbon content typically ranged between 200 and 2,000 mg C m^{-3}, with the highest values found at the inner site.

The spring phytoplankton bloom began in April, with chlorophyll levels remaining high (3 to 8 mg m^{-3}) at the inner sampling sites from May to the end of July, and a peak concentration of 12 mg m^{-3} was recorded in August. Lower concentrations were generally noted at outer stations and after August, a decline in concentrations to the winter minimum was observed throughout the Lough. Phytoplankton analysis showed that diatom species dominated the phytoplankton community throughout the growing season, and dinoflagellate species comprising no more than 5% of the total biomass at any time. Ball *et. al.* (1997) concluded that the higher summer chlorophyll levels, compared with data for the Irish Sea outside the Lough, suggest that high rates of remineralization were occurring within the Lough in the summer months. Tidal exchange represents a net loss of phytoplankton from Carlingford Lough without a compensation inflow of inorganic nutrients and Ferreira *et al* (1998) calculated that approximately 0.29 g N m^{-2} yr^{-1} as phytoplankton was exported to the Irish Sea. It was postulated that there is an upper level to the amount of bivalve production that the Lough can sustain. However, this upper limit is currently not exceeded and any future expansion which is constricted by the availability of suitable sites, would not approach a level which could be damaging to the Lough.

Carlingford Lough Aquaculture Association Ltd. (1990) reported that chlorophyll concentrations were low in comparison with other Irish Sea coastal sites at Killary Harbour and Dundalk Bay and it was suggested that the relatively large flushing effect in Carlingford may be responsible for exporting phytoplankton from the Lough. Higher chlorophyll *a* levels were observed at the shore sites, with the highest levels found at Greenore, Carlingford and Rostrevor and it was noted that all of these sites received untreated sewage. Also, higher levels were typically found in the west of the Lough, and may reflect an influence from the Newry River. Douglas (1992) found that primary production was greater along the centre of the Lough than along the northern shore.

To derive recommendations for the sewage treatment works in Newry, the extent of pollution in Carlingford Lough was assessed by the Department of Industrial and Forensic Science during 1975. Although water quality in the main water body was generally good, many sediments were found to be anoxic. The quality of the sediments deteriorated from the outer Lough to the tidal limit at Thompson’s Weir. On average, the organic content of the sediments was generally around 5% but rose to 10% in the inner Lough and at Warrenpoint Docks. In the river reach from the Docks to the tidal limit, the average organic content was approximately 12%, co-occurring with sporadic high sulphide concentrations. In the non-tidal reach, the sediment organic content of 15% and high sulphide content were related to the presence of industrial sources.

1.4 Eutrophication and EC legislation

1.4.1 Eutrophication

Eutrophication, defined as ‘the enrichment of water by nutrients, especially by compounds of nitrogen and phosphorus, causing an accelerated growth of algae and higher forms of plant life to produce an undesirable balance of organisms in the water concerned’ (EEC, 1991), is currently one of the main problems for fresh and coastal waters. To assess the trophic status of Northern Irish coastal waters in accordance with EC Directives, a number of criteria were defined by the DOE (NI) (1993) and include:

(1) winter nitrate-N levels
(2) Oxygen deficiency
(3) Changes in macrophyte growth
(4) Formation of algal scums on beaches and offshore
(5) Occurrence of exceptional algal blooms
(6) Changes in fauna
(7) Occurrence and magnitude of Paralytic Shellfish Poisoning (PSP).

1.4.2 EU legislation

Two European Union Directives are directly relevant to the control of eutrophication in the aquatic environment:

- The Urban Waste Water Treatment Directive (91/271/EEC) *‘to protect the environment from the adverse effects of urban waste water discharges and waste water discharges from certain industrial sectors’*.
- The Nitrates Directive (91/676/EEC) ‘*to reduce water pollution by nitrates from agricultural sources and to prevent further such pollution’*.

Urban Waste Water Treatment Directive

Under the UWWT Directive, member states are required to identify sensitive and less sensitive areas within fresh waters, estuaries and coastal waters. Sensitive areas are those which are eutrophic or which in the near future may become so if protective action is not taken. This identification of sensitive areas, together with the volume of the discharge to the receiving waters, has implications for the standard of sewage treatment required under the Directive, which establishes clear quantitative environmental standards with regard to nutrient levels. Under Article 4 of the Directive, secondary treatment is specified as the norm for waters identified as neither sensitive nor less sensitive. Less stringent treatment (but at least primary) is permitted for discharges to less sensitive areas. In the estuarine environment, this classification can only be applied where the population equivalent (PE) is less than 10,000 and where there are no existing or predicted eutrophication problems. Discharges from agglomerations with a population equivalent in excess of 10,000 into sensitive areas are subject to more stringent treatment. Where the population equivalent exceeds 100,000, the standard becomes more stringent and a maximum of 1 mg L^{-1} of total phosphorus and 10 mg L^{-1} of total nitrogen is imposed. Compliance with the above standards may require phosphorus and / or nitrogen removal within STWs, unless it can be demonstrated that nutrient removal will have no effect on the level of eutrophication.

Nitrates Directive

Nitrate from agricultural land is now the main source of nitrate in rivers and aquifers in Western Europe. Such pollution has human health risks, reflected in the setting of the European safety limit for nitrate in drinking water of 50 mg L^{-1} (Surface Water Abstraction Directive 75/440/EEC) and also implications for the whole aquatic ecosystem through eutrophication. Under the Nitrates Directive, member states must define estuarine, coastal and marine waters as polluted if they show signs of eutrophication, or if they may become eutrophic if action is not taken. In this context, only eutrophication induced by excess nitrates is relevant. Where a water body is so defined, its catchment must be designated as a 'nitrate vulnerable zone'. To meet the requirements of the Nitrates Directive, action programmes, consisting of mandatory measures, including the compulsory application of the Code of Good Agricultural Practice for the Protection of Water, must be adopted within these vulnerable areas.

The Code encompasses issues such as limits on application of organic manure (210 kg N ha^{-1} reducing to 170 kg N ha^{-1} after four years), adequate manure storage, matching inorganic N applications to the nitrogen requirements of the crops and maintenance of fertiliser usage records. At the time of writing, only two ‘nitrate vulnerable zones’ have been designated, namely at Comber in Co. Down and at Bellaghy in Co.Antrim, affecting approximately 1,250 hectares of agricultural land in total. More applicable to Carlingford Lough is the designation of the Mourne / Slieve Croob Environmentally Sensitive Area on the north shore of the Lough. The Code of Good Agricultural Practice must also be adhered to within this region.

The Countryside Management Division of the Department of Agriculture for Northern Ireland has been involved in trying to implement the Code of Good Agricultural Practice around the Newry Canal and in particular, the Jerrettspass area. The scheme which commenced in 1996 and will terminate during 1999, provided advice and assistance with making environment-based improvements to facilities and working practices (M. Carroll, DANI, Countryside Management Division, pers. comm.).

2.

Chapter 2. Nutrient inputs.

2. Nutrient Inputs

2.1 Introduction

In considering the stated aim of this project 'to identify existing or potential eutrophication problems in the Lough, and to make management recommendations in respect of any such problems', it is necessary to quantify and partition the incoming load of nutrients to the Lough. Quantifiable diffuse loads originate from agriculture, drainage from septic tanks, surface runoff and direct drainage. Atmospheric deposition of nutrients may be significant, particularly with respect to nitrogen. Point sources of nutrient enrichment include estuarine and coastal sewage treatment works and industrial sources. Locally significant nutrient concentrations may also be released from sediments, as end-products from biological and chemical pathways.

2.2 Methods

2.2.1 Sewage Treatment Works

Five sewage treatment works discharge directly to the Newry River estuary and Carlingford Lough (Fig. 2): Warrenpoint (population equivalent[1] of 13,887; secondary treatment), Cranfield (summer PE 4,000, winter PE 400; screened only) and discharges of raw sewage from Omeath (PE 255), Carlingford (PE 550) and Greenore (PE 143). A new secondary treatment works at Carlingford will be completed in 2000. The Newry works is of interest because of (a) its large size (PE 41,569; primary treatment) and (b) its outfall discharges downstream of this project's water quality monitoring site on the Newry River, therefore its nutrient load is not included in the Newry River budget. A new works, carrying out at least secondary treatment, will start construction soon.

[1] Population equivalent: 1 PE is the organic biodegradable load having a five-day biochemical oxygen demand (BOD5) of 60g of oxygen per day

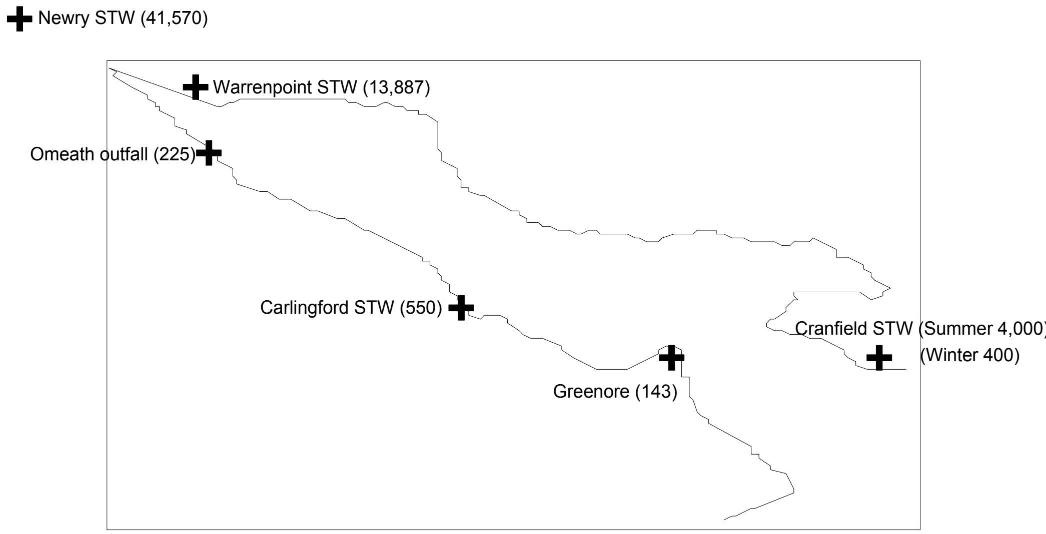

Figure 2.1 Map illustrating directly-discharging sewage treatment works

Only limited flow data for the Newry, Warrenpoint and Cranfield works were available from DOE (NI) Water Service. Flows were therefore based on population equivalents (PEs) on a *pro rata* basis, using average flows / PE from seven STWs in the Lough Foyle catchment. An adjustment was made for flows at the Cranfield works to account for the additional 3,600 PE during the summer months. This influx is attributed to holidaymakers staying in caravans. These temporary residents have a smaller nominal daily water consumption compared to residents living in permanent homes (180 litres and 230 litres, respectively).

DOE (NI) Water Service provided chemical data for Newry and Warrenpoint, collated on a weekly basis between 1994 and 1998 and for Cranfield on a monthly basis from May 1995. Nutrient loads were derived by multiplying flow and nutrient concentration. Monthly loads for Cranfield STW were calculated as the product of the load on the day for which chemical data were available (taken to represent an average daily load) and the number of days in the month. For the other two works, monthly loads were calculated as the product of an average of the four daily loads available for each month and the number of days in the month.

Neither flow data or chemical data for the Republic of Ireland's STWs were available from Louth County Council. Instead, loads were estimated on the basis of PE and the following estimates of nutrient load from crude sewage: 25 mg L^{-1} NH_3; 10 mg L^{-1} SRP; 0.25 mg $L^{-1}NO_3$ (A. Gibson, pers. comm.).

An estimate of the nutrient load from combined storm overflows (CSO) has not been included in the loading figures presented in this report. It is difficult to assess the number of CSOs and emergency overflows in the Warrenpoint system. There are 15 CSOs within the Newry sewage system and discharges during infrequent storm events have been estimated here. Since the model has not yet been run with rainfall data, it has not been possible to include a load estimate for this source in the current report. Although difficult to quantify their impact, the discharge of crude or screened sewage from CSOs may represent a locally significant source of nutrients.

2.2.2 Rivers and direct drainage

2.2.2.1 Flow data

Mean daily flow data for three gauging stations in the Newry River catchment i.e. Clanrye River at Mount Mill Bridge (station no. 206001; J086309), the Bessbrook River at Carnbane (station no. 206004; J074292) and the Jerretspass River at Jerretspass (station no. 206002; J064332), were provided by DANI Rivers Agency. The catchment areas to these gauging stations are 132.7 km^2, 34.5 km^2 and 32.4 km^2, respectively, summating to give a total gauged catchment area of 199.6 km^2. To estimate total flow at the bottom of the Newry River catchment (313.3 km^2), the sum of the flows at these three stations was scaled by a factor of 1.56. Flows from ungauged river catchments were calculated from the Newry River on a *pro rata* basis using catchment areas defined by using the Hydrologic modelling module which is part of the Spatial Analyst Extension of the ArcView 3.1 GIS package (see ArcView output in Figure 3). The sum of all these catchments within Carlingford Lough was 475 km^2.

2.2.2.2 Chemical analyses

Ten rivers flowing into Carlingford Lough were sampled on a bi-weekly basis during 1997 (Table 2.1).

Table 2.1 Position of river water quality monitoring sites

River	Irish Grid Reference
White Water	326 550 312 050
Cassy Water	323 850 314 950
Rostrevor River	318 100 318 350
Ghann River	317 550 318 250
Moygannon River	315 900 318 750
Newry River	312 600 319 400
Ryland River	314 650 315 900
Two Mile River	316 850 314 350
St. Patrick's River	318 000 313 350
Golden River	318 500 312 500

At each sampling station, field measurements of dissolved oxygen and water temperature were taken using a portable salinity-corrected meter (Model YSI 5739, Yellow Springs Inc., Ohio, USA). Water samples were filtered immediately, using Millipore prefilters and 0.45 µm membrane filters, into sterile containers. One unfiltered sample from each station was collected for Total Phosphate (TP) analysis. Samples were stored in the dark in cool boxes and returned to the laboratory, where they were frozen until analysed.

pH and conductivity measurements were taken in the laboratory [DANI Aquatic Systems Standard Operating Procedure LU/4.1 and LU/4.2, respectively]. Measurements of Soluble Reactive Phosphate (SRP), ammonium, nitrate and silicate were determined by Technicon Random Access Automated Chemistry System 800 4-channel continuous flow analyser, employing the methods described by Murphy and Riley (1962), Scheiner (1976); Downes (1978) and Golterman *et al* (1978), respectively [DANI Aquatic Systems Standard Operating Procedure BO/1131].

Concentrations of Total Phosphate (TP) were determined spectrophotometrically (Hitachi spectrophotometer V-2000) using the method described by Einsenreich *et al* (1975) [DANI Aquatic Systems Standard Operating Procedure LU/4.4]. Accuracy of methodology was monitored by participation in the National Marine Chemistry Analytical Quality Control Scheme (MPMMG, 1996).

2.2.2.3 Nutrient load estimation

Ammonium and SRP loads of the rivers were calculated by the method of Leopold and Maddock (1953, cf. Smith, 1976), where monthly loads are derived by plotting log concentration against log flow, to derive the constants for the following regression equation:

$$\log \text{load} = a + b \log \text{flow}$$

This has been shown to be the best methodology to calculate accurate and precise river loads from daily flow measurements and intermittent chemical monitoring data (Webb *et al*, 1997; Littlewood *et al*, 1998).

Nitrate loads cannot be represented by this relationship. Instead the nitrate loads were calculated by multiplying monthly total flows by flow weighted mean concentrations

$$\text{i.e.} \qquad \frac{\text{flow} * \text{concentration}}{\text{flow.}}$$

2.2.3 The atmosphere

The nutrient loading from the atmosphere was calculated from rainfall quality data collected at a monitoring site in Silent Valley (J306243), provided by Agri-Environment Group (DANI), where:

$$\text{load} = \text{concentration } (m^2) * \text{surface area of Carlingford Lough}$$

2.2.4 Industrial sources

Industrial consents data within the catchments of the Rivers Newry, Clanrye and Bessbrook was obtained from Environment and Heritage Service, Department of the

Environment. At present, only two consents which incorporate a condition relating to nutrient levels are in operation: these refer to two Council-run Landfill sites, situated at Irish Grid Reference J 125 292 and at J 145 250. Both consents set a maximum ammoniacal-N content for the effluent but only one stipulates a condition on the volume discharged (a maximum of 40 m^3 day^{-1}). Loads have therefore been calculated on the assumption that this volume applies to both sites, and using maximum permissible ammoniacal-N concentrations. The load from both of these sources is included within the Newry River total load, as both discharge points are further up the catchment than the position of the river quality monitoring site.

2.2.5 The Irish Sea

The Irish Sea's contribution to nutrient inputs in Carlingford Lough is difficult to estimate, as there is clearly an inward and outward flux of nutrients on each tide. The Lough has a Tidal Prism Volume of 126 x 10^6 m^3 which is similar to the low tide volume of 146 x 10^6 m^3. This large relative tidal exchange, together with generally higher nutrient concentrations inside the Lough than in the Irish Sea will mean that over an extended period, there is likely to be a net export of nutrients from the Lough to coastal waters (the effect of tidal dilution on nutrient and chlorophyll *a* levels is discussed in Chapter 3). The seasonal nutrient and chlorophyll *a* contour plots (Appendix 2) do not provide any evidence to suggest that coastal waters represent a significant source of nutrients. Therefore for the purposes of this report, it will be assumed that tidal exchange results in a net loss of nutrients from the system.

2.3 Results

2.3.1 Nitrate

During 1997, the total annual loading of dissolved inorganic nitrogen (DIN: NO_3^- + NH_4^+, of which < 2% was NH_4^+) to Carlingford Lough was calculated to be 1,311 tonnes. Approximately 77% (1,016 tonnes) of the DIN originated from the Newry River catchment, with an additional 11% from the other inflowing rivers (Fig. 2.2). The Newry and Moygannon River catchments have the highest nitrate areal loads within the Carlingford Lough area. These high loads are concurrent with a high percentage of land cover given over to arable farming and grassland (Table 2.2).

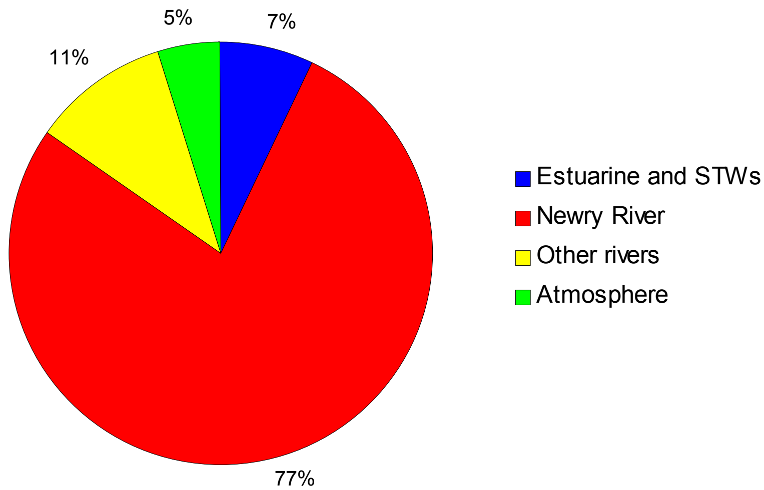

Figure 2.2 Partitioning of the total annual dissolved inorganic nitrogen (DIN) load

Table 2.2 1997 river and direct drainage areal loads (kg km^{-2} yr^{-1}) and land use for each catchment (CORINE data S McGuckin, pers. comm.).

River	NO_3	NH_4	SRP	Urban / industrial %	Arable + grass %
White Water	690	14	13	0.27	65.47
Cassy Water	397	7	5	0	35.15
Kilbroney River	731	7	80	2.26	28.62
Ghann River	1006	13	13	0.72	54.31
Moygannon River	2700	15	28	0	88.1
Newry River	3116	129	103	5.04	90.13
Ryland River	744	12	9	0	79.61
Two Mile River	610	4	4		
St Patrick's River	720	3	3	0	29.41

River nitrate loadings to Carlingford Lough are seasonal, with highest contributions during winter months (Fig. 2.3).

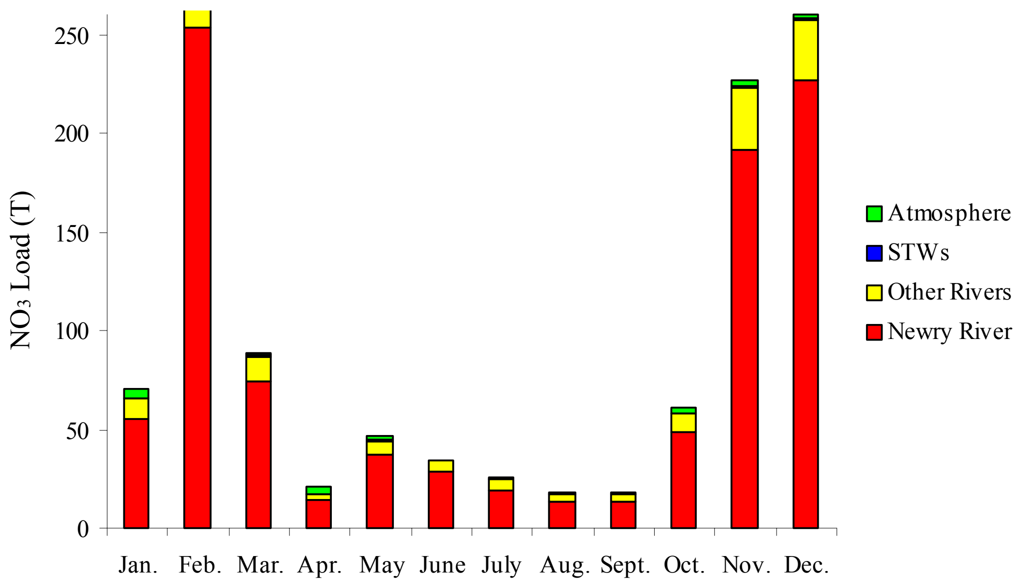

Figure 2.3 Monthly nitrate load partitioning

STWs make a negligible contribution (< 0.2%) to nitrate loadings to Carlingford Lough (Table 2.3).

Table 2.3 1997 total annual estuarine and directly-discharging sewage treatment works loads (tonnes).

Sewage treatment works	Nitrate	Ammonia	SRP
Newry	1.559	71.892	18.137
Warrenpoint	0.909	13.941	3.733
Cranfield	0.135	1.276	0.581
Omeath	0.007	0.646	0.259
Carlingford	0.012	1.184	0.474
Greenore	0.003	0.323	0.129

2.3.2 Ammonium

Although the atmosphere and Newry River are significant sources of ammonium to Carlingford Lough, the largest source is from STWs. Annual ammonium loads of 72 and 14 tonnes are discharged by Newry and Warrenpoint STWs, respectively, while the other 4 STWs together discharge 3.4 tonnes ammonium per annum (Table 2.3). Ammonium loadings are generally lower during the summer, where there is a significant reduction in the contribution from the Newry River (Fig. 2.4). Further, the atmospheric contribution is usually largest during months of high rainfall.

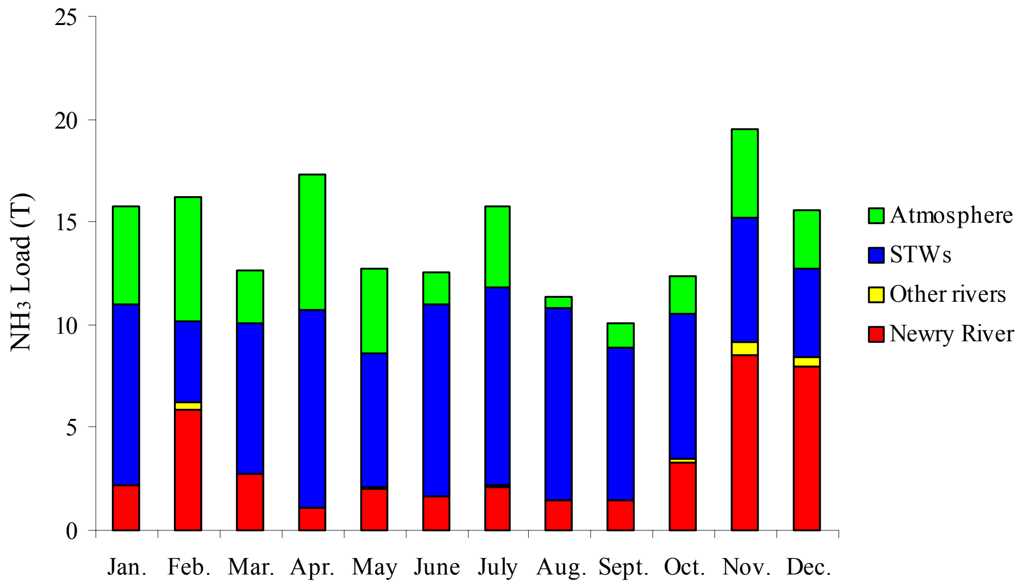

Figure 2.4 Monthly ammonium load partitioning

2.3.3 Soluble reactive phosphorus

The total annual SRP load to Carlingford Lough from estuarine, STWs, atmospheric and riverine sources is approximately 57 tonnes per annum. The two principal sources of SRP are from the Newry River (57%) and estuarine and STWs (38%) (Fig. 2.5). The Moygannon and Whitewater Rivers contribute 0.69 and 0.64 tonnes SRP per annum and the loadings for the other rivers are below 0.25 tonnes per annum. The majority of the STWs SRP load originates from Newry (18 tonnes) and a much smaller load from Warrenpoint (3.7 tonnes). The remaining STWs, taken together, contribute less than 2 tonnes SRP to Carlingford Lough (Table 2.3).

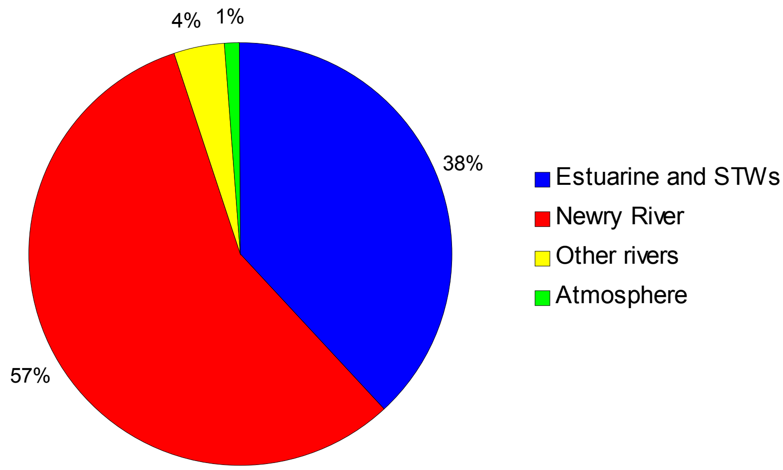

Figure 2.5 Partitioning of the total annual soluble reactive phosphorus (SRP) load

The relative contribution of STWs and rivers to the overall loading, is seasonal dependant (Fig. 2.6). During winter months, the Newry River represents the major source of SRP loading but in drier months, the estuarine and STWs become more dominant.

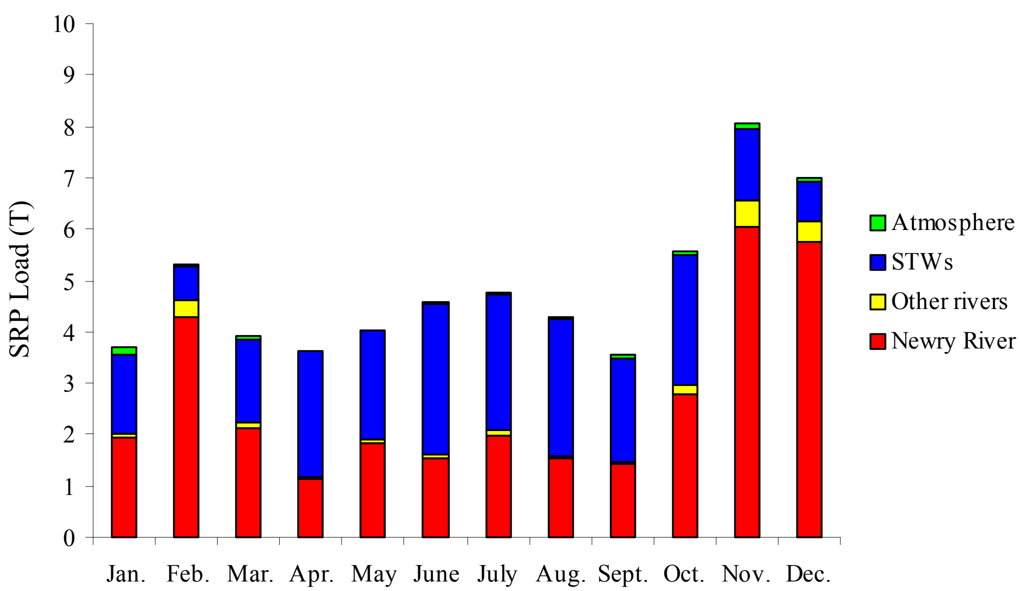

Figure 2.6 Monthly SRP load partitioning

2.4 Discussion

2.4.1 Rivers

Ball *et al* (1994) estimated that the Newry / Clanrye River was responsible for 70% of the total freshwater and freshwater-derived nutrient input into the Lough. Given that the loads of nitrate, ammonium and SRP from the Newry River, as a percentage of the total load are 84%, 24% and 57% respectively, it is clearly the single most important nutrient source in the Carlingford Lough system.

The Newry River has a high areal load for both SRP and DIN, reflecting both the high percentage of urban / industrial and arable farming / grassland CORINE land classes (Table 2.2). The Newry River receives effluent from a large urban population (even excluding the major works at Newry Town, which is considered separately) and is the site of major industrial activity in the region. Further, fertiliser application associated with crops and improved grassland will partially account for the high areal loads of SRP and DIN but complex factors such as soil structure and the existing N status of the soils must also be considered.

2.4.2 Sewage treatment works

Urban and industrial effluents are important sources of SRP and ammonium. Nevertheless, their contribution to nitrate loadings are negligible (Table 2.3). Nutrient loads in STW effluent are relatively constant throughout the year, although there is evidence of slightly higher loads of ammonia and SRP during the summer months. The largest works, at Newry, is by far the most important point source of all nutrients to the Lough, although a new works is currently under construction.

2.4.3 Atmospheric deposition

The atmospheric contribution of SRP and nitrate loadings to Carlingford Lough are insignificant, although as high as 38% of ammonium loadings may be atmospherically derived. A reduction in the atmospheric contribution to ammonium loadings is not possible on a local scale, instead is only achievable on a national scale.

Chapter 3. Monitoring of Carlingford Lough.

3. Monitoring of Carlingford Lough

3.1 Introduction

For the managing authorities, namely the Department of the Environment (NI) and Department of Agriculture, to meet the objectives under the Urban Waste Water Treatment and Nitrates Directives, it is necessary to monitor the current trophic status of waters covered by the legislation, and to identify any existing or potential adverse effects caused by anthropogenic nutrient loadings. This was the aim of the current study, which included an eighteen month estuarine and coastal monitoring programme. The Industrial Research and Technology Unit, on behalf of the Environment and Heritage Service, DOE, undertake a similar monitoring programme on an ongoing basis. This Estuarine and Coastal Waters Monitoring Programme is used to classify the estuaries around the coastline of Northern Ireland under the Estuaries Classification Scheme.

The Comprehensive Studies Task Team (MPMMG, 1997) outlines a range of standards to be used in defining adverse effects relating to waste water discharges. These include dissolved oxygen concentration, changes in benthic community structure, and predicted summer nutrient concentrations. Evidence of hypernutrification is not, however, an indication of adverse effects, since a region is potentially eutrophic only if the relative rate of light-controlled phytoplankton growth is greater than the relative water exchange rate plus the relative loss rate of phytoplankton by grazing. Therefore, adverse effects of primary productivity are defined by an assessment of the maximum chlorophyll *a* concentrations observed during monitoring. The DOE (1993), in laying down the methodology for identifying sensitive areas and vulnerable zones, also recognises the importance of water exchange in the development of eutrophication. This publication specifies that, in assessing the need to reduce nutrient inputs, consideration should be given to estuaries, bays and other coastal waters which have a poor water exchange.

3.2 Methods

3.2.1 Water sampling

Carlingford Lough was monitored for physical, chemical and biological parameters on 36 occasions between January 1997 and June 1998. Stations were selected to achieve maximum coverage of the Lough, within the constraints of boat access to the shallower areas (Fig. 3.1, Table 3.1).

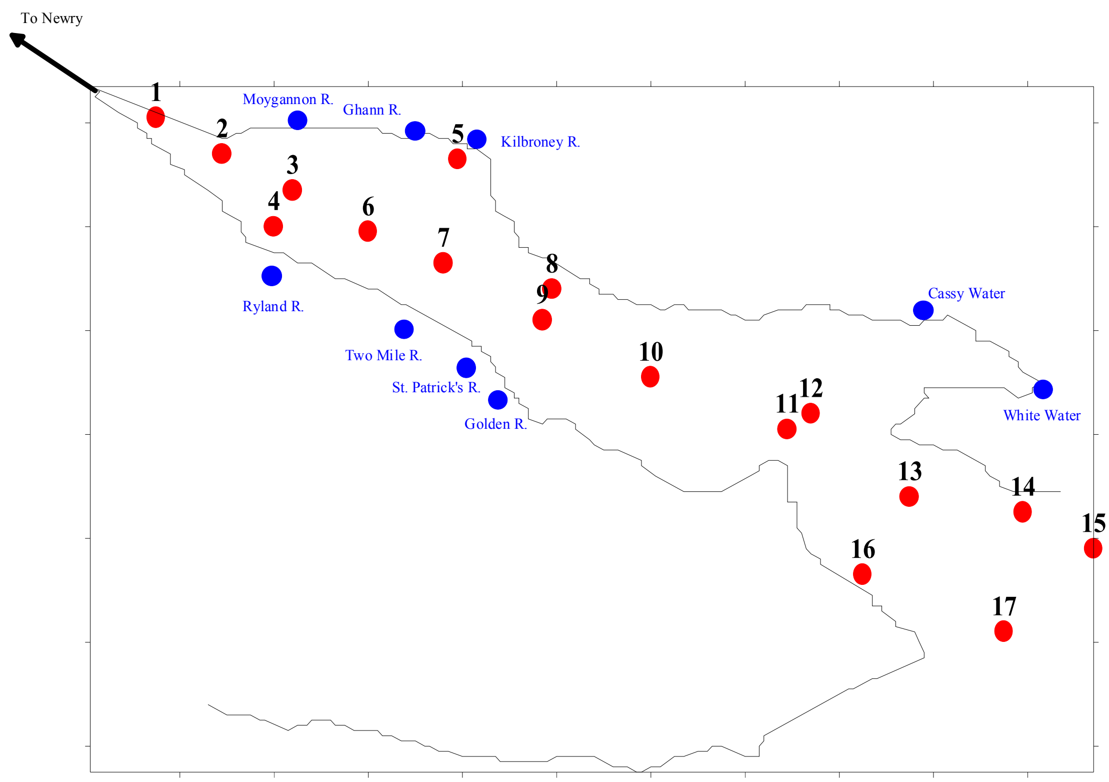

Figure 3.1 Carlingford Lough sampling sites.

Sampling was carried out monthly during the winter season of November to February, with weekly sampling during April and May, designed to maximise coverage over the period of the spring phytoplankton bloom. The Lough was monitored on a fortnightly basis between June - October. On each sampling occasion, all sites were sampled within five hours. sampling was carried out at all stages of the tidal cycle. Also, estuarine sampling was carried out at stations at Narrow Water and Victoria Lock, at the seaward end of the Newry Canal. This sampling was carried out between August 1997 and November 1998.

In situ measurements of physical parameters were made using a Seacat SBE 19-03 combined conductivity, temperature, depth (CTD) recorder, fitted with a Wet Labs Fluorimeter (Model 9601006). This instrument produced a detailed depth profile of salinity, temperature and fluorescence throughout the water column. Seasoft Data Acquisition Software Version 4.216 (Sea-Bird Electronics Inc., Washington, USA) was used to acquire and present the data. Light penetration was measured at each site using a 300mm-diameter Secchi disc.

Table 3.1 Sampling stations.

Site	Co-ordinates	
1	54 06 06	06 15 90
2	54 05 65	06 15 06
3	54 05 16	06 14 06
4	54 04 77	06 14 42
5	54 05 55	06 12 04
6	54 04 75	06 13 19
7	54 04 40	06 12 25
8	54 04 08	06 10 85
9	54 03 72	06 11 00
10	54 03 04	06 09 58
11	54 02 44	06 07 87
12	54 02 62	06 07 55
13	54 01 66	06 06 30
14	54 01 50	06 04 85
15	54 01 05	06 03 95
16	54 00 75	06 04 89
17	54 00 10	06 05 12

Water samples were collected at (a) one metre below the water surface, (b) at mid depth (as indicated by the vessel's echo-sounder), and (c) at one metre above the sediment, using a vertical water sampler. Unfiltered sub-samples were taken for chlorophyll *a* and phytoplankton community analysis and further unfiltered sub-samples were taken from surface samples, to quantify suspended particulate matter (SPM) and for elemental analysis of SPM. Remaining samples were immediately filtered through Millipore pre-filters and 0.45 μm membrane filters, to be used for soluble reactive phosphorus (SRP), silicate, ammonia and total oxidisible nitrogen (TON: nitrate + nitrite). All samples were stored in the dark in cool boxes and returned to the laboratory, where filtered samples were frozen until analysed, within one month of collection.

Measurements of Soluble Reactive Phosphate (SRP), ammonium, nitrate and silicate were determined by Technicon Random Access Automated Chemistry System 800 4-channel continuous flow analyser, employing the methods described by Murphy and Riley (1962), Scheiner (1976); Downes (1978) and Golterman *et al* (1978), respectively [DANI Aquatic systems Standard Operating Procedure BO/1131]. Chlorophyll *a* concentration was measured using the standard fluorescence technique described by Tett and Wallis (1978), including an acidification step to determine phaeopigment concentration [DANI Aquatic Systems Standard Operating Procedure BO/2122]. Analysis of phytoplankton communities at sites 1, 4, 9 and 16 were carried out on samples collected between January and November 1997, using the methodology of Utermöhl (1931, 1958) [DANI Aquatic Systems Standard Operating Procedure BO/2222]. Organic and inorganic SPM concentrations were determined by the methodology contained in DANI Aquatic Systems Standard Operating Procedure BO/2124. Elemental analysis of SPM, using the Carlo Erba NA1 500 Series 2 Elemental Analyser, followed the methods of DANI Aquatic Systems Standard Operating Procedure BO/1117. Accuracy of methodology was monitored by participation in the National Marine Chemistry Analytical Quality Control Scheme (MPMMG, 1996).

3.2.2 Sediment sampling

Sediment samples were collected at quarterly intervals between June 1997 and January 1998, for stations 1-10. Attempts were made to collect sediment from the remaining sites which lie seaward of site 10 but because of the nature of the substrate, it proved impossible. Grab samples were taken using a Van Veen grab. A core was taken from these samples, using a six centimetre-diameter coring tube, and then divided into three equal sections comprising the top six centimetres of the sediment. These sub-samples were analysed for total organic carbon / nitrogen content and for particle size analysis (see Charlesworth and Service 2000). A second core was taken using a 60 ml syringe, and this mini-core was divided into 2 centimetre sections, these being maintained in the dark for use in sediment chlorophyll *a* analysis.

The chlorophyll *a* content of sediments was analysed according to the methodology of Tett and Wallis (1978). The Forth River Purification Board performed particle size analysis, using a Malvern Laser Particle Analyser. Carbon and nitrogen contents of the sediments (using < 2 mm and < 63 µm fractions) were analysed using a Carlo Erba NA 1500 series 2 elemental analyser coupled to a stable isotope mass spectrometer (Europa Scientific Tracermass), and employing the DANI Aquatic Systems Standard Operating Procedure BO/1117).

3.3 Results

3.3.1 Sediments

As part of this project, sediments taken in June 1997 were analysed for grain-size characteristics, C^{org}, and metal concentration. The metal contamination status of the study area compared to other Northern Irish coastal sediments is discussed in Charlesworth and Service (2000). A brief description of the distribution within the study area shall be given here. Due to the coarse nature of the seabed only sites 1 – 10 were sampled. The grain-size analytical sheets are contained in the Excel database 'seds.xls' on the attached disc. Sediments range from very fine sand to medium silt, with > 80 % being < 63 µm except at three stations in mid-lough. Organic carbon also has a mid-Lough minima and highest concentrations at the head of the Lough reaching 3.2 % (Fig 3.2).

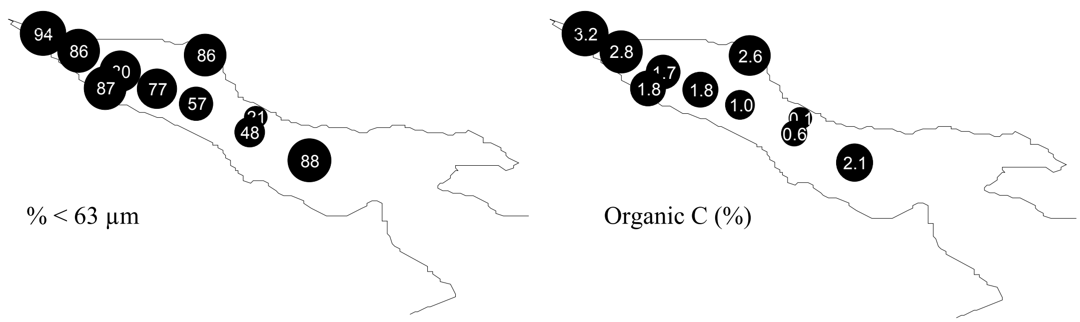

Figure 3.2 The distribution of silt and clay, and organic C.

All metals studied show highest concentrations in the area of fine sediments at the head of the Lough, and lowest concentrations in the coarse sediments of the mid-Lough. Exceptions to this include an anomalously high concentration of Hg (0.32 µg g^{-1}) near the head of the Lough, and a high concentration of Cr at the north shore of the mid-Lough. Aluminium, Fe, Cu, Ni, Pb and Zn correlate very well with sediment size, C^{org}, and between themselves (P < 0.01) indicating similar environmental controls, and no point contamination of metals (Table 3.2).

To eliminate grain-size effects (Loring, 1991) and therefore identify any contaminated sediments metal/Al ratios have been compared. Iron, Mn, Cu, Ni, Zn, and Pb, have comparable metal/Al ratios throughout the Lough with the exception of low ratios at sites 7 and 9, most probably caused by lithogenic variability. On the north shore of the mid-Lough a

Cr/Al ratio of 33 is recorded which is high in comparison to a maximum ratio of 19 in the remainder of the Lough. This result can be attributed to weathering of the underlying bedrock and/or the input of Cr from streams in this area which drain the metal rich Mourne Mountains, rather then inputs from an industrial or domestic origin. A high Hg concentration near the head of the Lough could be due to anthropogenic input, although the concentrations throughout the remainder of the Lough may be attributed to grain size.

Table 3.2 Correlation matrix of variables, n = 10 except for Pb and Hg (n = 8).

	Al	Cr	Cu	Fe	Hg	Mn	Ni	Pb	Zn	Corg	% S/C
Al	1	0.60	0.89	0.96		0.95	0.95	0.97	0.94	0.88	0.97
Cr		1	0.73	0.76		0.72	0.72	0.74	0.76	0.75	0.76
Cu			1	0.96		0.98	0.95	0.93	0.98	0.96	0.9
Fe				1		0.99	0.98	0.99	0.99	0.95	0.98
Hg											
Mn						1	0.99	0.99	0.99	0.95	0.96
Ni							1	0.98	0.99	0.94	0.95
Pb								1	0.98	0.85	0.88
Zn									1	0.97	0.96
Corg										1	0.92

3.3.2 Biological data

Diatoms dominated the Carlingford Lough phytoplankton community throughout the year (Table 3.3). Phytoplankton blooms, consisting primarily of *Chaetoceros* species, were recorded during spring and late summer / early autumn. In contrast, Douglas (1992) and Ball *et al* (1997), also working on Carlingford Lough, reported that *Thalassiosira* spp. constituted the dominant components of spring blooms. During the summer of the current study, the main diatom species present in Carlingford Lough included *Leptocylindrus* spp., *Nitzchia* spp., *Cerataulina pelagica*, *Thalassiosira* spp. and *Chaetoceros* spp. Douglas (1992) listed *Rhizolenia* spp. and *Chaetoceros* spp. as being dominant in summer samples whereas Ball *et al* (1997) reported that *Leptocylindrus* and *Asterionella* predominated under similar conditions. Both the findings of this current study as well as that of Douglas (1992), implied that *Chaetoceros* spp. was common in autumn samples in the current study. We also found high numbers of *Asterionella* spp. during this period.

Table 3.3 Phytoplankton species identified. (* denotes presence; + denotes present on ≥ 50% of sampling occasions).

Species	St. 1	St.4	St.9	St. 16	Mean cells L^{-1}
Diatom					
Amphiprora sp.		*		*	303
Asterionellopsis glacialis	+	+	*	*	2,760
Bacillaria paxillifera	*			*	95
Bellerochea malleus				*	4
Biddulphia alternans	+	*	*	*	162
Centric diatom	+	+	+	+	2,583
Cerataulina pelagica	*	+	*	*	4,220
Chaetoceros spp.	+	*	+	+	40,531
Corethron criophilum			*	*	7
*Coscinodiscus sp. **	+	*	*	*	106
Dactyliosolen fragilissimus	*	*	*		369
Distephanus sp.				*	19
Ditylum brightwellii	*		*	*	32
Eucampia zoodiacus	*		*	*	495
Fragilaria sp.	+	+	*	*	2,735
Fragilariopsis oceanica	*	*	*	+	1,033
Grammatophora sp.	*	*			38
Guinardia delicatula	*	*	*	+	785
Guinardia flaccida	*			*	8
Guinardia striata	*	*	*	+	108
Lauderia annulata			*		59
Leptocylindrus danicus	+	*	*	+	9,197
Leptocylindrus minimus	+	+	+	+	4,149
Lithodesmium undulatum	+	*	*	*	344
Lycmophora sp.			*	*	18
Melosira nummuloides	*			*	41
Navicula sp.	+	+	+	+	1,121
Nitzschia closterium	+	+	+	+	3,961
Nitzschia delicatissima	*	*		*	71
Nitzschia seriata	+	+	+	+	5,109
Nitzschia sp.	*				3
Odontella sp.	*	*		*	79
Paralia sulcata	+	+	+	+	384
Pennate diatom	+	*	+	+	943
Plagiogramma brockmannii	*				75
Pleurosigma sp.	+	+	+	*	1,086
Pseudo-Nitzschia sp.	*	*			25
Pyrophacus horologicum	+	*	*	*	168
Raphoneis sp.				*	4
Rhizosolenia faroense	*				41
Rhizosolenia fragillima	*				124
Rhizosolenia hebetata	*		*	*	427
Rhizosolenia setigera	+	+	+	+	338
Rhizosolenia sp.	*			*	22
Rhizosolenia styliformis	*		*		166
Skeletonema costatum	*	*	*	*	2,817
Striatella unipunctata				*	11
Thalassionema nitzschioides		*	*	*	70

Thalassiosira sp.	+	+	+	+	6,014
Dinoflagellates					
Ceratium furca			*	+	73
Ceratium fusus	*	+			20
Ceratium tripos	*		*	*	10
Unidentified dinoflagellate	*		*	*	148
Dinophysis sp.	*			*	73
Gonyaulax sp.	*				4
Gymnodinium sp.	*		*	*	54
Gymnodinium estuariale			*		4
Gyrodinium sp.	*	+	+	+	220
Heterocapsa sp.	*	*			57
Nematodinium sp.				*	2
Peridinium sp.	+	+	*	+	746
Prorocentrum micans	*		*	+	56
Triceratium alternans	*				19
Flagellate					
Unidentified flagellate	+	+	+	+	6,995
Silicoflagellate					
Dictyocha speculum	*			*	18

Dinoflagellate cells were not a major component of the Carlingford Lough phytoplankton community (present study) and represented less than 5% of the total biomass (Ball *et al*, 1997). Highest dinoflagellate numbers were recorded in June and it is believed that these were derived from the stratified waters of the western Irish Sea (Halligan *et al*, 1980, c.f. Ball *et al*, 1997). The most common dinoflagellates were *Gyrodinium* spp. which are more numerous between June and July and *Peridinum* spp. which thrive between March and August.

Varying levels of microflagellates were present throughout the year but were highest in late spring / early summer at approximately 25,000 cells L^{-1}.

Since 1993 DANI, in conjunction with the Department of Health and Social Security (DHSS), has been monitoring Carlingford Lough to detect the presence of toxic microalgae. Samples are taken at Killowen and Ballyedmund and to date, there has only been one occurrence of nuisance *Dinophysis* spp. bloom levels, and no toxicity was associated with it.

3.3.3 Water quality

3.3.3.1 Estuarine monitoring

The raw data are contained in the Access database 'Estuary All.mdb', on the attached disc.

Concentrations of the nutrients nitrate, SRP and silicate decreased sharply, moving from the freshwater site down the estuary (Table 3.4).

Table 3.4 Concentrations of chemical parameters at sampling stations in the Newry River estuary. The freshwater site at Newry Town is included for comparative purposes.

Para-meter	Newry River at Newry Town			Victoria Lock			Narrow Water		
	Min	**Mean**	Max	Min	**Mean**	Max	Min	**Mean**	Max
NH_4-N (µM)	0.44	**25.75**	89.12	1.32	**19.97**	58.83	0.84	**34.28**	96.55
NO_3-N (µM)	130.16	**276.59**	621.85	7.97	**142.01**	563.48	5.44	**84.93**	318.66
SRP-P (µM)	3.19	**7.53**	14.22	0.77	**2.84**	6.75	1.29	**3.64**	10.36
SiO_2-Si (µM)	56.72	**151.39**	301.86	7.27	**55.44**	157.12	9.17	**42.97**	88.98
Chl a (µg L^{-1})	1.00	**4.00**	8.00	1.00	**10.63**	34.00	1.00	**2.11**	5.00
SPM (mg L^{-1})	2.00	**33.50**	82.00	14.00	**72.00**	122.00	20.00	**59.00**	124.00

The spatial distribution of these nutrients is largely a function of loading patterns. Nitrate is sourced mainly from catchment runoff, with the Newry River representing the single largest source to the Lough. Similarly, silicate originates from rock weathering and enters the marine environment in river water. Approximately 57% of the total annual SRP load to Carlingford Lough is derived from the Newry River although in drier months, STWs are the major source, with Newry STW the predominant source. It is therefore logical that there should be a decreasing downstream gradient moving away from the primary source of these nutrients. Further, biological removal through primary production; sedimentation of phosphorus and conversion to phosphate minerals; denitrification of nitrate in the soft sediments of the estuary and also in the water column; and interaction of dissolved silicon with suspended material may also contribute to the spatial gradients reported. In particular, the extent of these physicochemical processes is often a function of temperature and salinity (Riley and Chester, 1971), which may change rapidly in the estuarine environment.

Ammonium is derived primarily from sewage treatment works and within the Carlingford Lough catchment, Newry and Warrenpoint STWs represent the major sources. These inputs probably explain the resulting concentrations found within the estuary.

Chlorophyll *a* concentrations in the present study were higher at Victoria Lock than at the other 2 stations. The critical value of 10 mg m^{-3}, constituted by the Urban Waste Water Treatment Directive for estuaries, was exceeded on 7 of 36 sampling occasions between May and October 1997.

Levels of suspended particulate matter were highest at Victoria Lock. Similar observations were recorded by DIFS (1976) and this was attributed to slippage of the mud covering the flood banks which protect the reclaimed land on the opposite side of the river from the canal. The DIFS (1976) study concluded that the suspended solids load in the estuary is the main influence on water quality, and there was strong evidence that the resuspension of anaerobic muds on the ebbing tide had an adverse effect on dissolved oxygen levels.

3.3.3.2 Carlingford Lough

Physical parameters

It is not practical to present all relevant data within this chapter therefore raw data may be found in the database 'Loughall.mdb' on the attached disk and further graphs and plots are included in the appendices.

Salinity

Salinity measurements are influenced by both tidal regime and freshwater input. As expected, mean salinity values increased from the estuary towards the Irish Sea (Table 3.5; Fig. 3.3a; Appendix 2). The minimum recorded salinity value of 26.24 psu was recorded at station 1 and the maximum of 34.87, at station 15, also the range within stations was greater at inner sites, reflecting the influence of freshwater inputs. In comparison to other Irish Sea Loughs, this salinity range is narrow and reflects that the flow from Newry River, which represents 70% of the total riverine flow into the Lough) is small in comparison to the tidal prism of Carlingford Lough : the average recorded daily flow volume of the Newry River is 470,707 m^3, compared with a tidal prism volume of approximately 146 x 10^6 m^3.

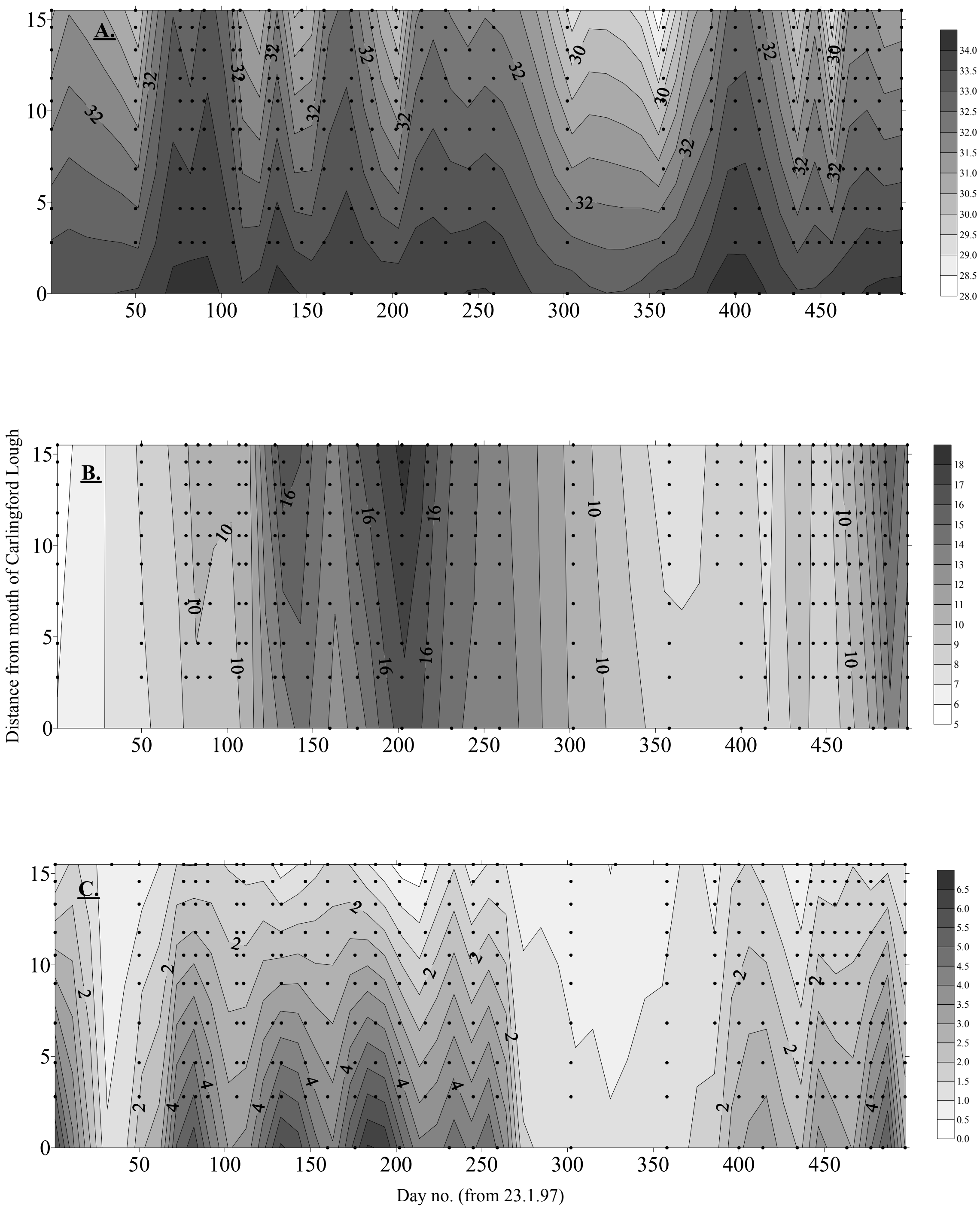

Figure 3.3 Carlingford Lough spatial plots:
A. Salinity B. Temperature C. Secchi depth

Table 3.5 Summary of physical data collected at marine sampling stations within Carlingford Lough.

Station	Salinity (psu)			Temperature (°C)			Secchi depth (m)		
	min	**mean**	max	min	**mean**	max	Min	**mean**	max
1	26.24	**31.31**	33.85	5.44	**11.98**	18.58	0.20	**0.88**	1.50
2	26.93	**32.28**	34.06	5.40	**11.97**	18.51	0.40	**1.17**	2.25
3	28.94	**32.80**	34.00	5.80	**11.91**	18.40	0.20	**1.57**	3.50
4	30.21	**32.62**	33.79	5.75	**11.78**	18.27	0.25	**1.49**	3.80
5	27.56	**32.52**	33.91	5.77	**11.62**	17.78	0.25	**2.01**	4.00
6	30.80	**33.09**	34.05	5.83	**11.84**	18.52	0.25	**2.00**	4.00
7	31.55	**33.28**	34.36	5.75	**11.77**	18.07	0.75	**2.35**	4.75
8	27.03	**32.99**	34.14	5.70	**11.46**	17.52	0.25	**2.45**	3.75
9	31.67	**33.42**	34.21	5.76	**11.50**	17.75	0.75	**2.55**	4.00
10	32.35	**33.54**	34.33	5.82	**11.49**	17.22	0.75	**3.06**	5.75
11	32.69	**33.76**	34.27	6.07	**11.55**	17.09	0.50	**3.53**	6.25
12	32.90	**33.86**	34.59	6.12	**11.53**	17.07	0.75	**3.80**	7.00
13	33.04	**33.91**	34.37	6.07	**11.37**	16.80	0.50	**3.75**	7.00
14	32.12	**33.99**	34.57	6.14	**11.44**	16.46	0.75	**4.01**	7.00
15	32.26	**33.99**	34.87	5.94	**11.31**	16.48	0.75	**4.24**	7.25
16	32.44	**33.96**	34.72	7.76	**11.65**	16.58	0.75	**4.16**	6.70
17	33.47	**33.99**	34.36	7.88	**11.86**	16.55	0.90	**3.31**	5.50

Temperature

The temperature profile within any sea lough is a function of the mixing of fresh and sea water and solar heating. Because the salinity gradient within Carlingford Lough is narrow, there was little variation between stations (Table 3.5; Fig. 3.3b; Appendix 2).

Stratification

Salinity and temperature data may be used to assess the degree of vertical stratification of the water column. Such stratification has implications for the availability of remineralized nutrients to the surface layer in summer, although in a shallow Lough such as Carlingford, this effect is less crucial. Pingree and Griffiths, 1978 (cf. Jenkinson, 1983) calculated a stratification parameter (S):

$S = \log_{10}(d) / |U|^3$ where, d is depth and $|U|^3$ is the cube of current speed (Ball *et al,* 1997) quote a maximum current speed near the lough mouth of 0.87 m sec^{-1} and in the inner lough, speeds regularly exceed 0.35 m sec^{-1} around the Rostrevor Narrows. DIFS (1976) quote an average velocity at Narrow Water of 0.39 m sec^{-1} for Spring tides and 0.30 m sec^{-1} for Neap tides).

Where S is >1.5 to 2.0, the water column is considered to be stratified. Based on the above figures, inner and mid sites in Carlingford Lough may be expected to experience some degree of stratification, but the water column at outer sites would be expected to be vertically homogeneous due to high current speeds and turbulent mixing reaching to the seabed. The greatest degree of both thermal and salinity stratification was recorded at the inner to mid sites (where maximum stratification was recorded in the winter months, this will have resulted from the outward movement of a colder, less saline layer of freshwater runoff, moved outwards from the estuary by wind). This finding is also illustrated in the depth plots of salinity and temperature (Appendix 4) which show a general increase in vertical homogeneity on moving seawards.

Secchi depth

Secchi depth measurements are indicative of the depth of light penetration. Within Carlingford Lough, secchi depth increased in the direction of the Irish Sea, from a minimum of 0.20 m at station 1 to a maximum of 7.25 m at station 15 (Table 3.5; Appendix 2) and was lowest during the winter months of October to February (Fig. 3.3c). Low secchi depth measurements recorded at the innermost stations probably reflects the high suspended matter loadings originating from Newry River and resuspensions of bed sediments in the shallower water depths from wind, wave and tidal actions. It is likely that some of this suspended load arises from the flood banks that protects the reclaimed land on the far side of the river from the canal (DIFS, 1976). Figure 3.4 illustrates the relationship between secchi depth and suspended particulate matter. At SPM concentrations greater than 50 mg L^{-1}, the secchi depth is less than 1.25 m whereas below SPM concentrations of 50 mg L^{-1}, the secchi depth is variable, indicating that other factors as well as SPM influence secchi depth.

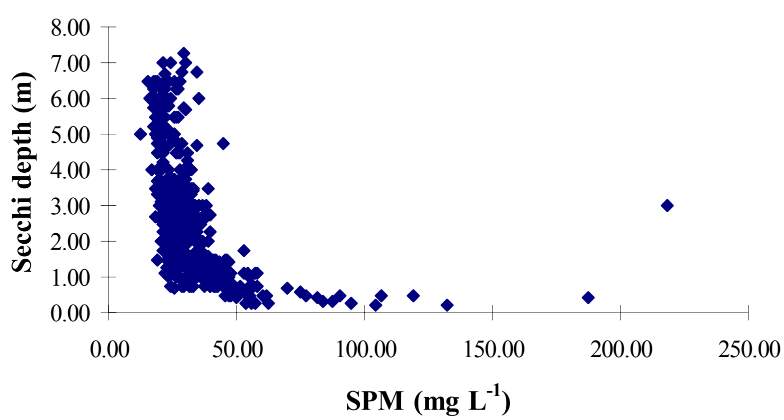

Figure 3.4 Relationship between secchi depth and SPM.

Secchi depth may be used to give an indication of the irradience, and by comparing this with the water depth, the potential for light to limit primary production may be evaluated. The compensation depth may be approximated from the depth of 1 % of the surface radiation (Tett, 1990) which may be estimated as being three times the depth of the secchi disc visibility (Parsons *et. al.* 1984). As Carlingford Lough is well vertically mixed, the compensation depth shown in Figure 3.5a may be thought of as a minimum value of potential light limitation, since gross production may be achieved below the compensation depth to the critical depth. The average total depths in the inner, mid, and outer lough at low tide are 3.6, 5.8, and 5.7 m. By comparison to Figure 3.5a, the compensation depth during the phytoplankton growing season is larger than the total depth therefore light does not generally limit phytoplankton growth.

Suspended Particulate Matter

Suspended particulate matter loads are a function of freshwater input, tidal range, salinity and wind stress (Dyer, 1972). Concentrations of SPM within Carlingford Lough were shown to decrease on moving seawards, from an average level of 46.76 mg L^{-1} at station 1 to an average of 26.02 mg L^{-1} at station 15 (Table 3.6).

Table 3.6 Summary of suspended particulate matter and organic SPM elemental composition data collected at marine sampling stations within Carlingford Lough.

Station	SPM (mg L^{-1})			SPM C:N		
	min	**mean**	max	Min	**mean**	max
1	23.60	**46.76**	132.67	5.23	**6.84**	9.31
2	24.30	**44.76**	187.40	3.31	**6.54**	8.95
3	20.28	**36.19**	104.66	4.12	**6.81**	10.31
4	22.80	**35.94**	87.80	3.56	**6.64**	9.66
5	21.43	**31.69**	61.60	5.10	**6.80**	8.60
6	18.86	**39.60**	218.57	4.76	**7.16**	11.14
7	18.29	**30.42**	47.20	4.78	**7.07**	10.90
8	19.42	**29.37**	57.68	4.25	**7.08**	10.64
9	16.86	**28.31**	43.60	3.95	**6.79**	10.02
10	19.00	**27.85**	54.85	4.31	**7.30**	17.88
11	16.10	**29.18**	55.00	4.11	**7.68**	16.18
12	16.20	**27.58**	58.40	3.93	**7.46**	28.63
13	18.25	**26.87**	60.56	4.05	**7.65**	23.21
14	17.15	**27.88**	58.04	4.57	**7.48**	14.50
15	12.29	**26.02**	47.00	2.09	**7.47**	12.64
16	15.43	**26.52**	51.60	1.08	**7.58**	15.11
17	18.85	**26.92**	42.50	1.52	**6.96**	10.93

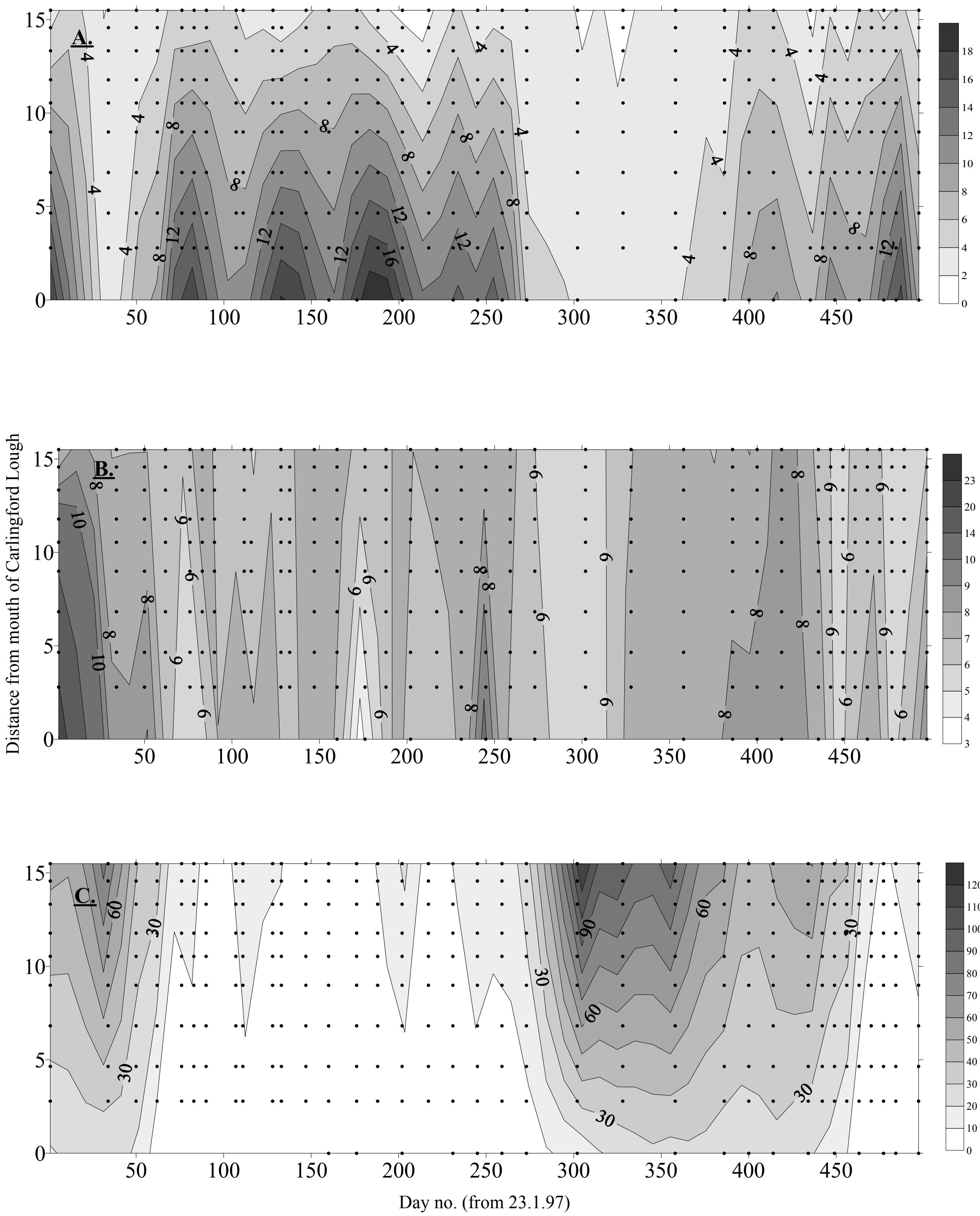

Figure 3.5 Carlingford Lough spatial plots:
A. Compensation depth B. C: N ratio C. Nitrate

This pattern may be explained by the influence of high SPM loadings from the Newry River together with the resuspension of sediment in the shallower inner sites from wind, wave and tidal action. Seasonal contour plots of SPM (Appendix 2) show higher levels during winter when river flow and storms are maximal and low levels during the summer when river flow is low. SPM concentrations from this study were generally between 30 mg L^{-1} and 50 mg L^{-1} and are within the range reported by Ball *et al* (1997).

Suspended particulate carbon and nitrogen (SPC and SPN)

The ratio of carbon to nitrogen in the suspended particulate matter can give an indication of the origin of the material. Phytoplankton have C/N ratios of between 3 and 6 approximately, and lithogenic material has a ratio of approximately 10 (Parsons *et. al.* 1984). Figure 3.5b shows that the C/N ratio varied temporally, with higher ratios of between 7 and 10 during the winter and lower ratios which were generally in the 6 – 8 range during the summer. This pattern reflects the higher contribution of lithogenic material in the winter due to higher SPM loads and resuspension of bed sediments arising from storm events and the higher contribution of phytoplankton biomass during the summer. Phytoplankton biomass is the predominant contributor to the SPM in the spring and summer and is probably the main source of food for the shellfisheries within the Lough at these times.

Nutrients

The Access database holding nutrient and chlorophyll data is to be found on the appended compact disc, named 'Loughall.mdb'.

Nitrate

In general, there was a significant reduction in the mean nitrate concentration and concentration range on moving seawards (Fig. 3.5c; Appendix 2). This pattern is commonly exhibited in estuaries, where the river at the head represents the major source of nutrient loading and the concentrations are affected by simple dilution effects and a combination of biological and abiological processes which both remove and supplement dissolved nutrients.

The spatial concentration gradient was particularly steep during winter (Fig. 3.5c; Appendix 2) and may be explained by high river flows during the wetter winter months and dilution of the riverine nutrient load on moving seawards. Winter nitrate concentrations are also augmented by biological processes e.g. nitrification, which is usually consummated by January (Riley and Chester, 1971). Further, wind direction also aids in establishing these horizontal gradients. Prevailing winds are from WSW to W and the most frequent gusts, associated with funnelling effects, come from between SW and NW therefore driving surface water seawards.

In temperate coastal waters, it is usual that high nitrate levels observed during winter fall rapidly in spring and remain at a low concentration throughout summer months when primary production becomes nutrient limited. A similar pattern was exhibited in Carlingford Lough where in mid – late summer, nitrate concentrations had dropped to low levels at all stations (Appendices 2 and 3). These minimum concentrations reflect biological uptake by phytoplankton and a significant reduction in nitrogenous inputs (Fig. 2.3) but may also be related to deposition of organic N in the sediments, biological uptake by submerged macrophytes and benthic diatoms as well as denitrification. It has been reported that in the soft sediments of the Lough, denitrification has the potential to remove significant quantities of nitrate, particularly during elevated summer temperatures (Livingstone, 1996).

By the beginning of August, negligible nitrate concentrations were recorded in the mid and outer regions of the Lough, while the influence of nutrient loading from Newry River is clearly illustrated in relatively higher concentrations at inner sites (Appendix 2). Seasonal depth plots of nitrate concentration (Appendix 4) illustrate the source of nitrate at the head of the Lough but also depicts the vertical homogenity of nitrate concentrations, reflecting effective mixing processes and the occurrence of primary production to the maximum depth. It is possible that primary production occurring on a large scale within the inner sites, will reduce the nitrate available for transportation to middle sites. At the outer sites, the effect of tidal dilution would be expected to be maximal, with a net loss of nutrients to the Irish Sea.

Ammonium

Within Carlingford Lough, average ammonium concentrations ranged from 1.57 mg L^{-1} at station 14 to 11.2 mg L^{-1} at station 1. At certain times, ammonium concentrations at stations 1 and 2 showed erratic fluctuations (Appendix 3) and it is likely that these reflect localised increases in ammonium loadings from the sewage treatment works at Warrenpoint or at Newry. The spatial plots for ammonium in Appendix 2 show that during the summer, ammonium concentrations are almost negligible in mid and outer sites. However, there appears to be an ammonium source around Carlingford and Greenore, presumably from the crude sewage. On the 12 August and 27 August 1997, ammonium concentrations were significantly reduced at inner sites while levels at mid and outer sites remained low. It is thought that this may be linked to the late-summer bloom (Appendix 2) and assimilation of ammonium (Ball *et al*, 1997). The winter ammonium spatial gradient was less distinct than for nitrate, reflecting the smaller contribution of ammonium loadings from freshwater runoff to ammonium concentrations within Carlingford Lough (Appendix 4).

Soluble reactive phosphorus (SRP)

During summer, SRP concentrations within the Lough were generally less than 1 µM. SRP depletion at 12 and 13 was in contrast to other surrounding sites and may be due to sinking of particulate phosphorus and loss to the soft sediments through conversion to mineral phosphates. The maximum level reported was 4.89 µM (at site 1) which is comparable with a value of 3.25 µM, recorded at Omeath by Ball *et al* (1997). On moving seawards, there was a decreasing trend in mean and range SRP values, with the highest mean concentration recorded at site 1. The STWs at Newry and Warrenpoint represent a major source of phosphorus loading to the Lough, and since both are located at the head of the Lough, there is a straight dilution gradient on moving seawards from these point sources (Appendix 2).

Silicate

Although on an annual basis, silicate levels decrease in a seawards direction, the seasonal distribution of silicate within Carlingford Lough is complex (Appendix 2). In general, maximum silicate concentrations were recorded in winter and minimum concentrations in summer. In early spring, silicate concentrations were generally highest at outer sites but during summer, the pattern was reversed, with highest concentrations at inner sites and concentrations seawards of Rostrevor Bay were extremely low. This distribution during the growing season reflects both the status of freshwater runoff as the main source of silicate, as

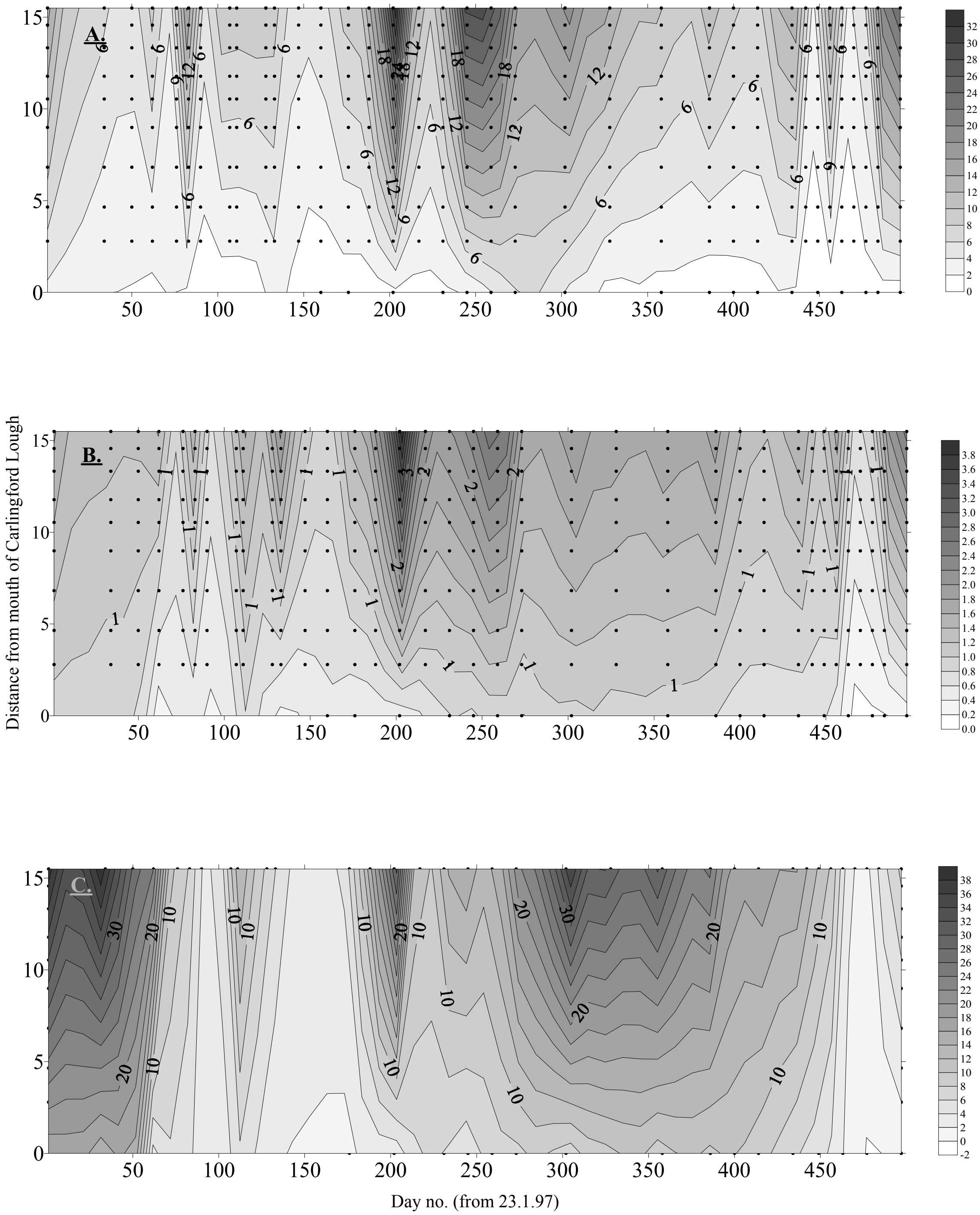

Figure 3.6 Carlingford Lough spatial plots:
A. Ammonium B. Soluble reactive phosphorus C. Silicate

well as its utilisation by the diatom-dominated phytoplankton community within Carlingford Lough. By late August, silicate concentrations seawards of site 7 had started to rise again, while concentrations at all other sites remained low. It is possible that the higher silicate concentrations may be associated with the inverse relationship with chlorophyll *a* concentrations (Appendix 2)**,** caused by the absence of a late summer bloom which was noted at all other sites within the Lough. During winter, when there was no phytoplankton blooms, a significant decrease in silicate concentrations in a seawards direction was recorded, reflecting that the main source for this nutrient is the Newry River.

Chlorophyll a

Mean chlorophyll *a* concentrations within Carlingford Lough decreased on moving from the inner to outer sites (Appendix 2)**.** The highest chlorophyll *a* concentrations of 11.29 µg L^{-1} and 12.40 µg L^{-1} were recorded at sites 2 and 4, respectively. It is probable that productivity was highest at these two sites because of locally elevated nutrient levels, arising from their proximity to the inflow of the Newry and Ryland rivers, respectively. It is important to note that although nutrient levels at 1 are higher than those at 2, the light regime at 2 may be more favourable for phytoplankton production.

The spring bloom was detected on the same date in both years throughout the Lough. Ball *et al* (1997) noted that the timing of the spring bloom in Carlingford Lough is usually delayed by a few weeks with respect to other sea loughs around the Irish coast, but occurs earlier than in the neighbouring Irish Sea. This has been attributed to a tidal dilution effect.

One of the most interesting features of the chlorophyll *a* spatial plots is that after the spring phytoplankton bloom, the summer primary production fell at sites seaward of Rostrevor Bay, however production at sites landwards of Rostrevor Bay continued at a high rate. Ball *et al* (1997) had reported a similar finding and an explanation may centre around release of nutrients from the sediments during summer months. During August, the inner and mid sites experienced a second significant phytoplankton bloom which was not a feature of outer sites. Chlorophyll *a* concentrations were uniformly low during the winter months.

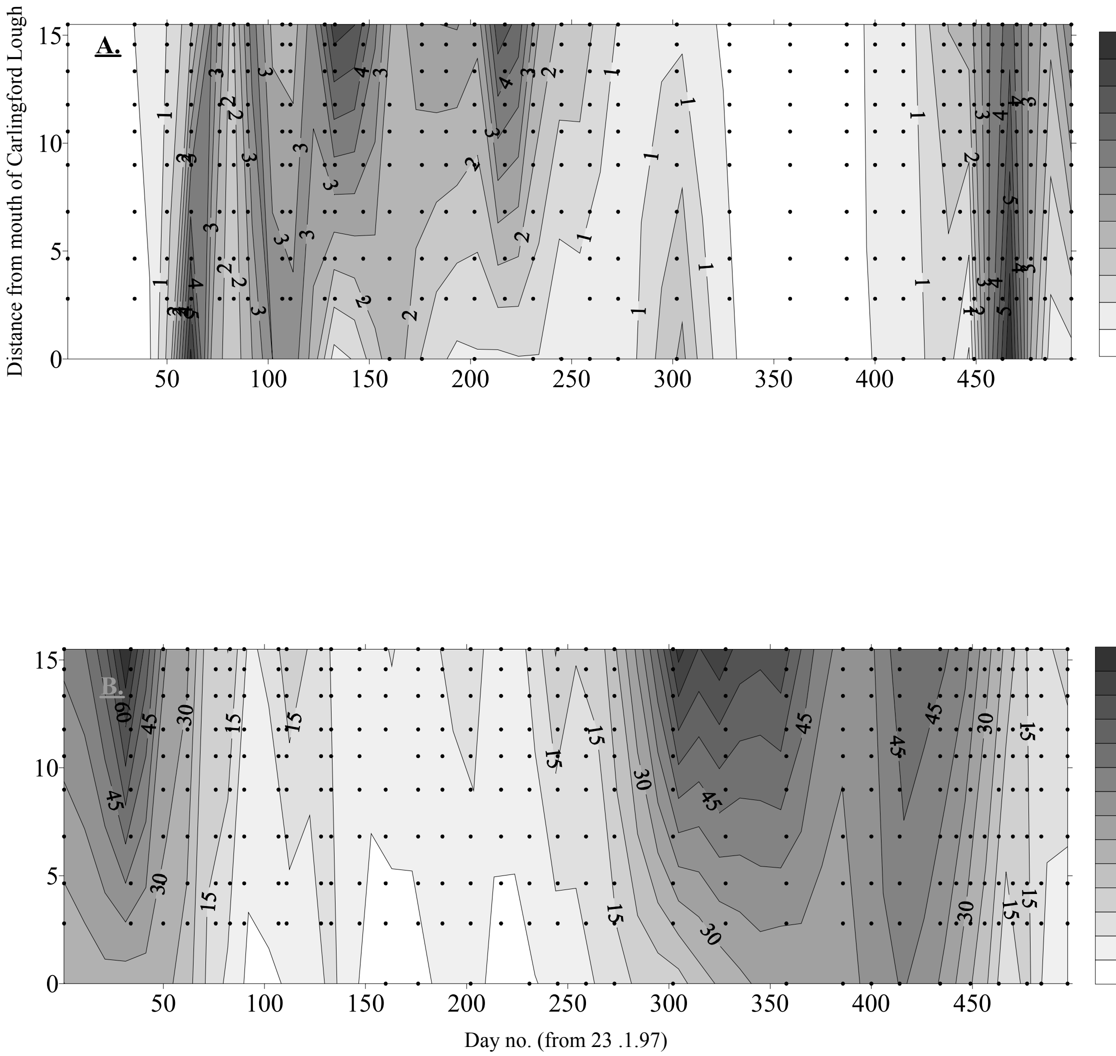

Figure 3.7 Carlingford Lough spatial plots:
A. Chlorophyll a B. N:P ratio

3.4 Discussion

In Northern temperate regions, the single most important limiting factor triggering the spring bloom is day length, followed by water temperature. In early spring, the availability of nutrients is adequate to support an increasing phytoplankton population. However, results from the current study as well as those of Ball *et al* (1997), IRTU (1995) and Carlingford Lough Aquaculture Association Ltd (1990), clearly establish nitrogen availability as the main environmental variable limiting phytoplankton growth during summer months.

The theoretical optimum value of the nitrogen : phosphorus ratio with respect to phytoplankton growth is usually quoted as 15:1 (Redfield, 1934). However, there is evidence that a lower ratio of 10:1 is more applicable to shallow coastal waters (Ryther and Dunstan, 1971). In the current study it was assumed that when the ratio fell below 10, dissolved inorganic nitrogen availability was limiting to phytoplankton growth. By mid-April, all but the two most inner sites were developing signs of nitrogen limitation (Fig. 3.7). The extent and duration of nitrogen limitation was demonstrated to increase on moving seawards and at the outer sites, nitrogen availability limited productivity right into October, when day lengths again precluded maximal phytoplankton growth (Appendix 4). This was also reflected by summer chlorophyll *a* levels which did not drop to a summer minimum at the inner sites, while seaward sites experienced a significant reduction following the initial bloom.

High chlorophyll *a* concentrations during August were recorded at all sites landwards of station 10 (Appendix 3) but productivity remained low at outer sites. The corresponding chlorophyll *a* concentrations showed highly significant negative correlations with dissolved inorganic nitrogen and N:P ratios at inner and mid sites (Table 3.7).

Table 3.7 Correlation coefficients of chlorophyll *a* concentration *v*. nutrient concentrations and water transparency by Lough zone

August	Secchi depth	[DIN]	[SRP-P]	[Silicate-Si]	N:P
Inner	-0.1338	-0.7461	-0.7639	-0.5387	-0.8453
Mid	-0.7767	-0.7148	0.4435	-0.4480	-0.7506
Outer	-0.1086	-0.0069	-0.8188	-0.0.41	-0.3439

(Inner zone = sites 1 – 6 Mid zone = sites 7 to 12 Outer zone = sites 13 to 17)

Following the spring phytoplankton bloom, concentrations of chl *a* at sites 10 - 16 fell to low levels, whereas concentrations at stations 1- 10 were sustained. It is possible that the concentrations in the inner and mid lough were able to be maintained due to a continuous source of nutrients to this area. Much of the nutrients are discharged to the inner Lough via riverine sources but these are generally low during these time periods. It is therefore possible that nutrient regeneration within the sediments and water column during times of low direct inputs are sufficient to allow primary production to continue at a rate comparable to the spring bloom found in Belfast Lough (Livingstone and Smith, 1999). The outer Lough may not show such an effect due to its nutrient status being dominated by that in the Irish Sea. Without further study it is difficult to differentiate the between the importance of direct vs diffuse nutrient inputs.

SRP limitation in Carlingford Lough is unlikely to be a limiting algal growth factor as algae require phosphorus in smaller quantities than nitrogen and it is regenerated from organic matter more quickly than nitrogen. Similarly, silicate levels fluctuated within Carlingford Lough but is of importance only where it exerts a limit on phytoplankton growth in the presence of excess nitrogen, therefore silicate limitation does not pertain in Carlingford Lough.

In certain loughs, light penetration may be limiting to primary production. Due to the shallow nature of Carlingford Lough and the relatively small river flow during the drier months, light is not thought to be an important controlling factor. Also, turbulence and salinity are not important limiting factors in Carlingford Lough.

Carlingford Lough Aquaculture Association (1990) reported that primary production within Carlingford Lough may actually be higher than indicated by chlorophyll *a* measurements, with the deficit attributable to zooplanktonic grazing. Shellfish mariculture operations and wild mussel beds in the Lough will undoubtedly have an effect on the standing stock of phytoplankton cells, although this has not been quantified. Ball *et al* (1997) addressed the impacts of shellfisheries on Carlingford Lough and concluded that the role of ammonification in maintaining summer chlorophyll *a* levels places an upper limit on bivalve production, but that this limit has not yet been reached. Production is further limited by the net loss of phytoplankton to the Irish Sea, without a compensating tidal load of inorganic nutrients. A comparison of the residence time and clearance time of the current shellfish stock showed that the clearance is by far the greater, indicating that a considerable increase in biomass could occur before the clearance time : residence time ratio becomes critical. It can therefore be assumed that the impact of shellfish on chlorophyll *a* levels in the Lough will be spatially localised, and the impact on the ecosystem as a whole will be small.

Primary productivity within Carlingford Lough is greater and the spring bloom, 2 – 3 weeks earlier than in the adjacent Irish Sea (Ball *et al*, 1997). Higher productivity within the Lough may partly reflect a more favourable relationship between critical depth and mixing depth within the Lough than in the deeper waters of the Irish Sea. Ball *et al* (1997) postulated that the large relative tidal prism volume of the Lough (190 x 10^6 m^3 cf. 195 x 10^6 m^3 low tide volume) may indicate that the net growth rate represents a balance between growth and exchange rates. Dilution of the Lough's waters occurs mainly through tidal exchange and may limit *in situ* phytoplankton growth (Gowen *et al,* 1983). Tidal dilution effects may also be partly accountable for the delay in the timing of the spring bloom, with respect to other areas around the coast of Ireland (Ball *et al*, 1997), and for the lower chlorophyll *a* levels recorded in the Lough, compared to neighbouring Dundalk Bay (Douglas, 1985. cf. Ball *et al*, 1997). This is investigated in the following chapter.

Chapter 4. Mathematical modelling.

4.1 Introduction

From a study of the movement of water in Carlingford Lough by mathematical modelling the residence times of water within 'boxes' may be computed and hence a greater understanding of the dynamics and behaviour of physical, chemical and biological parameters may be gained. In particular it may be used to determine the susceptibility of a box to bloom events given that other limiting factors are favourable.

4.2. Methods

Carlingford Lough is generally well mixed (Ball *et. al.* 1997; Chapter 3), and the freshwater input small compared to the large volume and tidal range. The tidal regime is the primary determinant for mixing within the lough, and therefore a segmented tidal prism model has been applied. The method follows that of Ketchum (1951), and Dyer, (1972), however is modified to exclude the effects of freshwater input on the mixing processes. During summer typical freshwater input are < 2 m^3 s^{-1} which contributes less than 0.001 % to the low tide volume of the Lough day^{-1}. A modification of this sort is therefore deemed acceptable. This allows the placement of the most landward box according to tidal currents recorded at Warrenpoint (Anon., 1993), so it's length equals the average distance travelled by a particle on the flood tide. Let this first box have a low tide volume of $\mathbf{V_o}$ with an intertidal (tidal prism) volume of $\mathbf{P_0}$. The limit of the next box is placed so that:

$$\mathbf{V_1 = V_0 + P_0} \qquad \mathbf{(1)}$$

Therefore the low tide volume in each box equals the total prism within the next box to landward, plus the low tide volume in box **0** or:

$$\mathbf{V_n = V_0 + \Sigma n\text{-}1 \text{ to } 1P} \qquad \mathbf{(2)}$$

Each box contains, at high tide, the volume of water contained in the next seaward box at low tide. Thus the limits of the boxes, are equal to the average excursion of a particle during the flood tide. If the mixing within each box at high tide is assumed to be complete then the proportion of water removed on the ebb tide is the ratio between the local intertidal volume and the high tide volume. Thus an exchange ratio can be defined for each box as:

$$\mathbf{r_n = P_n/(P_n + V_n)} \qquad \mathbf{(3)}$$

The flushing time in tidal cycles for each box will be $\mathbf{1/r_n}$. The total flushing time for the lough will be the sum of the flushing times for the separate boxes. Volumes were calculated at high and low water (4.63 and 1.2 m above chart datum (chart 2800)) at mid neap and spring at 1 or 2 m depth interval contours, and are believed to be accurate to within 5 %.

Average DIN concentrations at all depths at each station within a box were averaged for winter (January and February) and summer (July and August). Station 1 has not been considered as it is an estuarine site and has DIN concentrations that are not representative of concentrations in box 1. Concentrations in the North West Irish Sea have been taken as 6 and 1.5 µM DIN for winter and summer respectively (B. Stewert *pers. comm*.). Inputs of DIN were also calculated for the same period from the Newry River, Newry, and Warrenpoint STWs, and atmospheric sources. The other smaller STWs, and rivers have not been considered as their contribution to the nutrient load is small (Chapter 2), and within the noise of the model.

From the above the potential, steady state, nutrient concentrations can then be estimated (CSTT, 1994) by:

$$\mathbf{S = S_o + ((s_i)/(E.V)} \qquad \mathbf{(4)}$$

Where:

- $\mathbf{S}$ is the concentration of DIN in the box under consideration.
- $\mathbf{S_o}$ is the concentration of DIN in the adjacent water
- $\mathbf{s_i}$ is the total of inputs to the box under consideration
- $\mathbf{E}$ is the exchange rate with adjacent water
- $\mathbf{V}$ is the volume of the box under consideration at mid-tide.

Averaged over a tidal cycle, water within any box is conserved, therefore for boxes 2 and 3 $\mathbf{S_0}$ has been taken as the average of DIN concentrations in adjacent boxes.

4.3. Results and Discussion

The placement of boxes, the considered inputs, volume and flushing times can be seen in Fig 4.1 and Table 4.1.

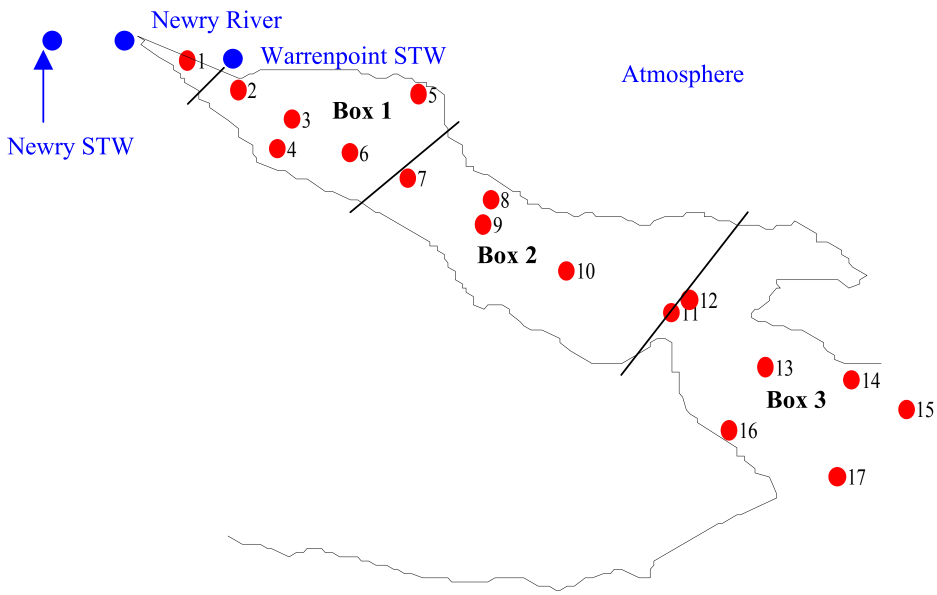

Figure 4.1 Boxes, considered inputs and stations used for model.

Table 4.1 Characteristics of boxes used for model.

	Box 1	Box 2	Box 3	Whole Lough
Stations	1 - 6	7 - 10	11 - 17	
Surface Area (km^2)	9.0	13.3	14.9	37.2
Mean low tide depth (m)	2.9	4.3	4.2	3.9
Low tide volume ($\mathbf{V_n}$) (10^6 m^3)	26.3	57.0	63.1	146.4
Tidal prism volume ($\mathbf{P_n}$) (10^6 m^3)	30.7	45.1	50.6	126.4
Exchange ratio ($\mathbf{r_n}$)	0.54	0.44	0.45	0.46
Flushing time (hrs)	22	27	27	76
Mean winter DIN inputs (10^3 mol day^{-1})	398.4	9.2	10.4	418
Mean summer DIN inputs (10^3 mol day^{-1})	68.4	2.8	3.2	74

Using equation 2 to define the limits of box 3 positions the seaward limits outside of Carlingford Lough. As the seaward limits of Carlingford Lough are well defined and mixing and currents outside the Lough different, the limits of this box have been taken as from Cranfield to Ballagan Points. The low tide volume of this box therefore does not equal the high tide volume of Box 2, and the flushing time of the whole of the Lough is likely to be approximately 10 % less.

Table 4.1 shows that the residence time of each box is approximately a day, and of the whole Lough 3 days. Greater than 90 % of the input of nutrients are discharged to Box 1 and summer loads are < 20 % of winter predominantly due to the influence of the Newry River. The observed, predicted from current loads, and predicted loads following an increase of loads from the STWs, and riverine sources by 20 and 50 % are shown for winter and summer in Tables 4.2 and 4.3 respectively.

Table 4.2 Modelling results for winter DIN concentrations.

DIN concentrations	Box 1	Box 2	Box 3
Observed	53.9	41.8	27.5
Predicted	52.4	40.8	24.0
Predicted + 20 %	54.5	40.8	24.0
Predicted + 50 %	57.7	40.9	24.1

Table 4.3 Modelling results for summer DIN concentrations.

DIN concentrations	Box 1	Box 2	Box 3
Observed	5.6	2.6	2.2
Predicted	5.3	3.9	2.1
Predicted + 20 %	5.9	3.9	2.1
Predicted + 50 %	6.7	3.9	2.1

The predicted and observed DIN concentrations correlate well in winter when internal processes are at a minimum. During the summer predicted concentrations are higher than observed in box 2 indicating a loss within the system probably due to biological uptake. Because of the short flushing time and large volume of box 1, inputs are quickly dispersed, so that by increasing inputs concentrations a concurrent increase in DIN in the water column is not apparent. Therefore by increasing inputs to box 1 by 50 % only a 10 % increase in water column concentrations is observed. Predicted concentrations in boxes 2 and 3 are a function of mixing with adjacent water, and therefore largely controlled by concentrations in box 1 and the N-W Irish Sea.

To estimate the possible effects of increasing DIN inputs on chlorophyll 'a' concentration, regressions were performed between chlorophyll 'a' and DIN on all dates during spring and summer according to the method of Gowen *et. al.* (1992). Of the 24 dates studied only 4 showed negative relationships with an r^2 of > 0.3; of these only 2 showed an r^2 of > 0.5 (Table 4.4).

Table 4.4 Relationship between chl 'a' and DIN during spring and summer.

Date	µg chl (µmol N)$^{-1}$	r^2
25/03/97	0.26	0.66
15/04/97	0.18	0.33
12/08/97	2.19	0.36
27/08/97	3.46	0.52

The poor relationships on each date at all stations show that factors other than DIN must also be important in determining chl *a* concentrations. However, chapter 3 showed that DIN was the most likely factor limiting phytoplankton growth, and furthermore, Table 3.7 showed that good relationships do exist between the two variables if stations are grouped synoptically and analysed on a seasonal basis. It is therefore difficult to accurately predict the increase in chl *a* following an increase in DIN concentrations. However, using the median value of 1.05 µg chl l^{-1} produced from 1 µM of NO_3 (Gowen *et. al.* 1992), a 50 % increase in DIN inputs to Box 1 during the summer would be expected to elevate chl *a* concentrations from 0.2 µg L^{-1} to 1.2 µg L^{-1}.

Ferreira *et. al.* (1998) implemented an ecological model of Carlingford Lough which revealed that biological processes were far more important that physical exchange in controlling phytoplankton mass fluxes (Table 4.5). Those authors suggested that natural mortality contributed approximately 80 % of phytoplankton losses presumably by zooplankton grazing and sedimentation. As chl concentrations are so low in the outer Lough dispersion out of the area of consideration probably accounts for little loss of the total standing crop. Assuming a mean intrinsic growth rate of 0.27 day^{-1} of phytoplankton (Gowen *et. al.,* 1983), and the flushing time of the whole lough to be less than 3 days, suggests that phytoplankton populations do not remain within the Lough long enough to double in concentration. Chlorophyll is generally < 5 $\mu g\ l^{-1}$ in the outer Lough, thus concentrations in the mid Lough would be expected to be < 10 $\mu g\ l^{-1}$ due to physical constraints upon phytoplankton growth if it was not limited by nutrients and light, which is generally the case.

Table 4.5 Mass balance for phytoplankton in Carlingford (gN m^{-2} yr^{-1}), calculated from model simulation (from Ferreira *et. al.* 1998).

Physical		Biological	
Inputs		Sources	
Upstream	0.04	Net 1° production	1.94
Outputs		Sinks	
Downstream	-0.33	Natural mortality	-1.58
		Oyster filtering	-0.07

In summary, because of the large volume to input ratio and large dispersive capacity of inner Carlingford Lough, any increase in DIN inputs of < 50 % are unlikely to cause significant increases of DIN concentrations in the water column. Although DIN appears to be the primary factor limiting algal biomass, other biological and physical constraints are also important. Therefore it is difficult to predict the effect of increasing DIN concentrations on chlorophyll levels.

Chapter 5. Summary.

5. Summary and conclusions

5.1 Nutrient inputs and behaviour

Direct inputs of nutrients are predominantly to the inner Lough. On an annual basis the Newry River contributes 84, 24, and 57 % of the total loads of NO_3, NH_4 and SRP respectively, and is clearly the most important source of nutrients. STWs are the largest source of NH_4 although ammonium contributes < 2 % of the total load of DIN. The load of NO_3 is seasonal with the load in winter months typically being > 100 T $month^{-1}$, but during the summer loads are generally reduced to < 50 T $month^{-1}$. There is little variation in the load of SRP and NH_4 $month^{-1}$ over the course of a year due to the partitioning of the load between other sources that do not show such seasonal influence.

In general nutrient concentrations in the Lough decreased on moving seawards, due to the major sources being at the head of the Lough, and subsequent dilution with nutrient poor water from the Irish Sea. Nutrients showed the characteristic seasonal patterns of summer minima and winter maxima found in temperate waters determined by variations in inputs and biogeochemical processes. Following the initial spring bloom of phytoplankton, NO_3 concentrations were generally reduced to < 1 µM in the mid and outer Lough until September and the N:P ratio fell below 15. At some stations in the inner Lough concentrations of NO_3 were higher than the mid and outer during the summer probably due to a combination of the nutrient sources discharging to this zone and remineralization of nutrients in the shallow water and sediments.

5.2 Limitations to algal biomass

Throughout the Lough chlorophyll concentrations increased from winter minimum concentrations to between 5 and 10 µg l^{-1} during March and April. Following this initial 'bloom', the stations could be broadly grouped into two according to the trends observed. Seaward of station 10 chlorophyll concentrations decreased to < 3 µg l^{-1}, whereas concentrations were sustained at > 3 µg l^{-1} at stations landward of station 10. The evidence presented suggests that N is the primary limiting factor for phytoplankton growth, and therefore the sustained levels of chlorophyll in the inner Lough are probably caused by continued inputs of nutrients to this zone, either directly and/or by recycling within the water column and sediments. Other biological and physical constraints are also likely to be important limiting factors to phytoplankton biomass.

5.3 Trophic status and compliance with EC directives

The Comprehensive Studies Task Team (MPMMG, 1997) states that a region is eutrophic if observed chlorophyll *a* concentrations regularly exceed 10 $\mu g\ l^{-1}$ during the summer. During this study, the critical chlorophyll concentration of 10 $\mu g\ l^{-1}$ was exceeded at stations 2, 12 and 13 on one occasion and more than once at stations 4 and 7 although levels never exceeded 12 $\mu g\ l^{-1}$. There is no other evidence to suggest that the Lough is detrimentally affected by anthropogenic discharges or activities and can not therefore be considered eutrophic.

5.4 Recommendations

Further research should be carried out to elucidate the importance between direct inputs of nutrients and inputs via biogeochemical processes in the water column and sediments.

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